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**Accounting for the Relationship of Agricultural Land
to Economic and Ecological Welfare: Conceptual
Framework and Case Study in the Grand
River Basin of Ontario, Canada**

by

Terence McRae (B. A., B. Sc.)

A Thesis submitted to the Faculty of Graduate Studies and Research in
partial fulfillment of the requirements for the degree of

Master of Arts

Department of Geography and Environmental Studies

Carleton University

Ottawa, Ontario, Canada

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Thesis Supervisor

Chair, Department of Geography and Environmental Studies

Carleton University

September 2000

Abstract

This thesis develops and implements an environmental accounting framework for agricultural land which takes fuller account of its relationship to economic and ecological welfare. An aspect of this framework is applied to Ontario's Grand River basin for 1981, 1991 and 1996. The application assesses stocks of soil on cropland, flows of soil into water, and resulting economic impacts on water users. The findings suggest soil losses into water do not threaten soil productivity, but imposed economic costs of some \$9.5 million on basin water users in 1996.

This study suggests environmental accounting offers a broader appreciation of how agricultural lands can affect welfare. Limitations of environmental accounting include the sensitivity of the economic results to the valuation methods used. Additional research would be required to develop accounts for other resources and materials, improve monetary valuation of results, and estimate the costs of using agricultural land in ways that reduce externalities.

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provided the basic data on agricultural land use and management for the case study of the Grand River watershed, and prepared the maps which show the study area and the results of the calculations. Peter provided data on the clay content of soils in the watershed.

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Part A: Introduction

1. Issue Statement.

Over the past two hundred years, two key factors have together brought humanity into the present, unprecedented era of high economic activity. The first of these, the scientific and technological revolution (which is still underway), has led to both increased levels and new forms of economic activity (WCED, 1987). The second factor (in part a consequence of the first) is the explosive increase in human populations, made possible through advances in agricultural productivity, medicine and trade (WCED, 1987).

Economic activity is closely linked with the physical environment and the aggregate economy is part of the larger natural ecosystem (Daly, 1991). For example, environmental resource goods (such as minerals, land and water) are used as inputs into economic production processes, and environmental services (such as nutrient cycling) help assimilate and recycle wastes from the economy. This general relationship between the environment and the economy has always been in place but, as pointed out by Daly (1991), recent changes in the scale of economic activity have fundamentally altered aspects of the relationship. For most of human history, the scarcity of environmental goods and services was not an economic constraint as their availability exceeded the requirements of the human economy. Thus, in aggregate, the marginal user cost of resource use was essentially zero and the presence of externalities was not a pressing issue. In the present period of high human demand for environmental goods and services, demand may exceed supply and some forms of economic activity may deplete or degrade the underlying base of natural capital. In other words, current use of natural capital may

diminish future opportunities. Tradeoffs are thus a frequent reality, and policy makers are faced with the challenge of balancing economic, environmental and social concerns in order to maximize current welfare and ensure sustainability for future generations (WCED, 1987). The term *sustainable development* has been coined to express this challenge (WCED, 1987).

Information is needed to both guide and evaluate policy-making for sustainable development. For most of the past century, policy has been largely oriented to achieving economic growth. Consequently, systems to generate information for policy purposes (of which the leading example is the System of National Accounts - SNA) have primarily focused on transactions in the market economy. However, the SNA has been criticized for its failure to account for the use, appreciation or depreciation of natural capital due to economic activity (see section 5.2). If the leading indicators used by policy-makers are silent on the environmental aspects of economic development, policy failure (in which economic policy goals are pursued at the expense of other policy objectives, such as environmental quality) may result, possibly leading to unsustainable forms of economic development and sub-optimal levels of current and future welfare. This holds true for all sectors of the economy, including resource sectors such as agriculture, which use stocks of natural capital (such as soils and biological diversity) in production and interact with other aspects of the natural world, such as with climatic systems and aquatic ecosystems.

Clearly, if more sustainable economic development is to be achieved, we must have some idea of whether the path we are on is headed toward or away from this goal. Adjusting or supplementing the SNA and other policy-support information systems to

take into account market failures related to natural capital is a key challenge facing analysts, and an important step towards improving decisions so that more sustainable forms of development are achieved.

2. Thesis Objectives and Structure.

The objectives of this thesis are to:

- Develop a conceptual framework for analyzing the relationship between agricultural land and economic and ecological welfare using an environmental accounting approach.
- Apply selected aspects of this framework to agricultural land in a specific area of Canada using existing physical and economic data.
- Draw conclusions about the utility of environmental accounting methodology for assessing the relationship between agricultural land and overall welfare.

The structure of the thesis report is as follows. Part B reviews alternative concepts of sustainable development and selected frameworks developed by various authors for its assessment. Of these, particular treatment is accorded the environmental accounting approach, with illustrative examples presented of how this framework has been applied to the natural resources sectors of land, agriculture and forestry. Part C describes in general terms the linkages between primary agriculture and the environment, and develops a conceptual framework for assessing linkages between agricultural land and welfare based on environmental accounting principles. In Part D, an aspect of this framework is applied to agricultural land in a study area of Ontario. Part E presents the overall conclusions of this study.

Part B: Recent Developments Regarding the Analysis of Sustainable Development.

This section provides an overview of various perspectives on what sustainable development is (chapter 3), reviews operational frameworks developed to measure its achievement in society (chapters 4 and 5), and presents examples of how one specific framework, environmental accounting, has been applied to its measurement in the natural resource sectors of land, agriculture and forestry (chapter 6). Chapter 7 discusses issues which remain outstanding concerning the use of environmental accounting for measuring sustainable development, and chapter 8 provides concluding comments about the approaches reviewed here.

3. Concepts and Definitions of Sustainable Development.

The term *sustainable development* was popularized by the World Commission on Environment and Development (WCED) in its landmark publication “Our Common Future” (WCED, 1987). The WCED defined this concept as “*development that meets the needs of the present without compromising the capacity of future generations to meet their needs*”. The concept of sustainable development was subsequently accepted by governments, industry and others at the 1992 UN Conference on Environment and Development. Official endorsement did not, however, end the debate. Many analysts criticized it as being so general as to be virtually meaningless (see, for example, Lohman, 1990 and Rees, 1990). Others pointed to a need for further definition to give it more practical meaning and to make it operational.

The debate about the meaning of sustainable development is complex. Colby

(1991), in exploring the sustainable development discourse, identified five underlying paradigms of sustainability (Table 1). These provide a reference frame from which to situate and understand the underlying philosophical positions of various authors, studies and interests. Each paradigm holds different assumptions about human nature, the natural world and the interaction between the two, as well as different ways of perceiving evidence, threats, risks and benefits.

Colby (1991) describes the “frontier (or classical) economics” paradigm as viewing nature as having a virtually unlimited supply of physical resources (e.g., raw materials, energy, etc) to be used for human benefit, and also as having a virtually unlimited capacity to assimilate the by-products (e.g., waste residuals) of the production and consumption of outputs. At the opposite end, Colby (1991) describes the “deep ecology” paradigm as advocating integration of some of the more scientific aspects of systems ecology with a “biocentric” view of the relationship between humans and nature. Among its basic tenets are species equality, reductions in human populations, and non-growth economies. The remaining three paradigms are situated between the extremes of frontier economics and deep ecology, and exhibit various characteristics of each.

Table 1: Alternative Paradigms of Sustainability (Colby, 1991)					
Paradigm	Frontier Economics	Environmental Protection	Resource Management	Eco-Development	Deep Ecology
Dominant theme	"Progress" as infinite growth	"Tradeoffs" as in ecology vs economic growth	"Sustainability" as necessary constraint to growth	Co-develop humans & nature; re-define security	Eco-topia, anti-growth, harmony with nature
Human-nature relationship	Very strong anthropocentric	Strong anthropocentric	Modified anthropocentric	Ecocentric	Biocentric
Dominant threats	Hunger, poverty, disease, natural disasters	Human health impacts of pollution	Resource degradation, population growth	Ecological uncertainty, global change	Ecosystem collapse
Main themes	Open access, resource exploitation	Remedial / defensive, legalize ecology	Global efficiency, economize ecology	Ecologize economy, restructuring	Back to nature, biospecies equality
Who pays?	Property owners, public	Income tax payers	Polluter pays	Pollution prevention pays	Avoid costs, forego development
Strategies	Industrial agriculture, fossil energy, unregulated waste disposal, free markets	End-of-pipe cleanup, market regulation, some prohibition, environ'tal assessment	Impact assessment, risk management, eco-efficiency, conservation strategy	Uncertainty management, eco-technology, low-input agriculture	Stability management, reduce scale of market economy, intrinsic values

O’Riordan (1989) offers another, related perspective (Table 2). He argues all societies hold a somewhat schizophrenic view toward the natural world, a duality which consists of a “nature-as-usufruct” view and a “nature-as-nurture” perspective. The former paradigm, which O’Riordan suggests is the dominant view in all industrial societies, views nature as malleable and manipulatable through the ingenuity of scientific understanding and technological applications. Underlying this paradigm is a strong belief in the capacity of humanity to improve welfare through managerial ability. The nature-as-nurture view, according to O’Riordan, arose as an antidote to the nature-as-usufruct perspective. This view is more cautious, pessimistic and critical. O’Riordan suggests the “environmental” movement of today exhibits tensions between these two views.

Evidence of the different paradigms articulated by both Colby (1991) and O’Riordan (1989) is widespread, and underscores the ambiguity which underlies the concept of sustainable development. For example, at/near the “deep ecology or “ecocentric” end of the scale are those who hold that view that economic development and environmental quality are incompatible. Proponents emphasize limits in the Earth’s biophysical carrying capacity and question the value of growth itself, as evidenced by the report of Meadows et al., (1972) “*The limits to growth*”, which suggested that prevailing trends in human populations, industrialization, pollution and resource depletion, if continued, would lead to uncontrollable declines in human welfare within 100 years. Similarly, other analysts, such as Daly (1993), suggest the concept of growth be replaced by a steady-state (and, they argue, more sustainable) economy.

Table 2: Perspectives on Environment and Resource Management (O'Riordan, 1989).				
Ecocentrism			Technocentrism	
<u>Gaianism</u> Faith in the rights of nature and of the essential need for co-evolution of human and natural ethics	<u>Communalism</u> Faith in the cooperative capabilities of societies to establish self-reliant communities based on renewable resource use and appropriate technologies		<u>Accommodation</u> Faith in the adaptability of institutions and approaches to assess, evaluate and accommodate environmental demands	<u>Intervention</u> Faith in the application of science, market forces and managerial ingenuity
Green supporters, radical philosophers	Radical socialists, committed youth, radical liberal politicians, intellectual environmentalists		Middle-ranking executives, environmental scientists, white-collar trade unions, liberal-socialists politicians	Business and finance managers, skilled workers, self-employed, right-wing politicians, career-focused youth
0.1% of various public opinion surveys	5-10% of various public opinion surveys		55-70% of various public opinion surveys	10-35% of various public opinion surveys
Demand for re-distribution of power towards a decentralized, federated economy with more emphasis on informal economic and social transactions and the pursuit of participatory justice			Belief in the retention of the status quo in the existing structure of power, but as a demand for more responsiveness and accountability in political, regulatory, planning and educational institutions	

The concept of biophysical limits or constraints to growth is also reflected in the writings of Erlich and Erlich (1993) and Erlich et al., (1993), who point out limits and constraints imposed by the first and second laws of thermodynamics. Similarly, Hubbert (1993) suggests the present period of exponential growth is an anomaly from a past characterized by a much smaller population and a low level of technology, energy consumption, dependence on non-renewable resources and very slow rates of change. He then suggests the future must also be characterized by slow rates of change and utilization of renewable energy sources capable of sustaining a human population of optimum size at a comfortable standard of living for a prolonged period of time. Others more representative of the “technocentric” or “frontier economic” view advocate solutions based on continued economic growth.

The sustainable development concept has also received much attention in the discipline of economics. Standard economic theory assumes that labour and capital are the sole factors of production (Binswanger, 1998). However, many economists have expanded this view to other factors, including the environment. One early example is Juster et al (1981), who suggested that societal resources are limited by two basic factors: the amount of available human time and the stock of wealth inherited from the past. This wealth stock they broadly defined as the sum of conventional, tangible capital assets, intangible human and other capital assets, stocks of organizational capital (e.g. networks of social support systems) and stocks of environmental assets. Similarly, Pearce and Turner (1990) expand the concept of capital beyond its traditional meaning in economics to also encompass natural capital, which they defined as the capacity of the environment

to supply resources and assimilate wastes. From this, they logically suggest that a sustainable economy is one that maintains a constant stock of natural capital over time, and propose the following sustainability rules or criteria:

- Renewable resources are to be used in such a way that the harvest rate is not greater than the natural regeneration rate.
- As non-renewable resources are depleted, their reduced stock must be compensated for by increases in renewable resources
- Waste flows to the environment should be at or below the assimilative capacity of the environment.

Generally speaking then, an “ecological” branch of economics has emerged which seeks to recognize the value of natural capital to economic development and welfare. An ongoing debate remains, however, concerning the substitutability of different forms of capital. For example, Hartwick (1978) and Solow (1986) represent a mainstream view which recognizes environmental goods and services as capital, but which suggests that unless there are good reasons for singling out one form of capital, the requirement for sustainable development is that subsequent generations receive an aggregate stock of capital at least as great as what exists now. The key implication here is the notion that the various types of capital (produced capital, social capital and natural capital) are substitutable one for another, a view Pearce and Turner (1990) call “weak sustainability”. Others hold the view that in addition to maintaining the total level of capital intact, some attention should be given to the composition of that capital, and that the stocks of these different forms of capital must be maintained separately, a view Pearce and Turner (1990)

call “strong sustainability”.

However, some economists continue to harbour reservations about using the term “natural capital, and Hinterburger et al., (1997) review some of the critiques offered. Victor (1991) points out there is no single theory of capital to which economists subscribe and examines three schools of thought: mainstream neo-classicism, the “London school” (e.g., Pearce and others), and the “thermodynamic school” (e.g., Daly, Georgescu-Roegan). Victor points out that an essential feature of capital is that it is reproducible by human action, which is not the case with respect to the environment. Hinterburger et al., (1997) agree with this view, point out other problems concerning valuation of natural capital (reviewed in chapter 7) and urge caution in using the term capital for the environment. England (1998) also raises concerns about this term, arguing that ecological economists will waste much time should they try to vigorously pursue refinement and measurement of so broad and diverse a concept. He instead urges that the notion of natural capital be retained as a powerful metaphor, but that effort be focused on developing a rigorous measure of sustainable economic welfare.

In conclusion, very differing views exist about the meaning of sustainable development. These can be grouped according to the alternative paradigms identified by Colby (1991) and O’Riordan (1989), or through the notions of strong versus weak sustainability advanced by Pearce and Turner (1990). The concept of sustainable development as advanced by the WCED held the hope of compromise: economic growth was presented as the solution to both environment and development concerns provided efforts were made to achieve a higher quality economic growth. It seems evident that, at

least for now, this more mainstream technocentric paradigm has dominated economic reform in the developed world and economic development in the developing world. Virtually all governments have chosen economic growth as their core strategy for enhancing welfare, coupled with attempts to make this growth more sustainable. The adherents of Colby's "deep ecology" and "ecodevelopment" paradigms have largely been relegated to the margins of mainstream policy-making, and much of the ongoing debate is focused on defining sustainability within a growth context. Nonetheless, the debate remains a complex and value-laden one. A single and unifying theory of sustainable development has yet to emerge, a fact which complicates its measurement immensely.

4. Frameworks for Measuring Sustainable Development.

A framework is a conceptual model that can help identify and provide insights into the issues, relationships and linkages among the elements of a system under consideration. The lack of an accepted definition of sustainable development has not stopped governments and other public and private institutions from promoting sustainable development (as they define it) and not surprisingly, a variety of frameworks for assessing it have emerged. Several of these are reviewed below, drawing on the classification developed by Hardi et al., (1997) and Friend and Rapport (1991).

4.1 Stress-Response Frameworks.

These macro-level frameworks have been widely used in the sustainable development measurement literature and all versions stem from the initial work of Rapport and Friend (1979). The basic elements of the framework are as follows:

- a) Human activities give rise to stressors which affect the environment, of which the

key ones are: resource harvesting/depletion, generation of waste residuals and physical restructuring of the environment and landscape

- b) These stressors may lead to changes in environmental conditions of quality and quantity
- c) Collective and individual responses may be initiated to manage and reduce stresses.

The stress-response model is based on the notion of a causal relationship between stress-generating human activities and changes in the natural environment. The model suggests that with adequate responses, environmental stresses can be controlled and impacts mitigated or prevented. Related models, such as the OECD Pressure-State Response model (OECD, 1999b; Figure 1) and the CSD Driving Force-State-Response Model (www.un.org/esa/sustdev/worklist.htm), are examples of similar frameworks.

A strength of these frameworks is their utility for organizing bio-physical environmental data and statistics in a manner which reflects potential causal relationships at the macro level. The main limitations, discussed by both Bassell (1999) and Alfieri (1999) are twofold. First, their general nature -- they neglect the systematic and dynamic nature of the processes they describe, they cannot fully account for interactions between framework elements at the micro level (although they were not designed for this purpose), and they may suggest causality where none exists. Second, they were not conceived as tools for economic analysis of the environment, and therefore cannot incorporate monetary data, or provide direct links to economic variables, thus limiting their ability to identify economic and environmental interactions.

Source: OECD, 1998b.

4.2 Capital-Based Frameworks

Capital-based frameworks have their roots in economics where measurement systems such as the System of National Accounts were established to measure produced wealth and capital accumulation. This notion is extended to sustainable development, which is viewed as a process of managing wealth, or capital, of which there are two types in addition to produced capital: social/human capital and environment/natural resource capital. Social and human capital refers to the set of rules, institutions and trust within societies that allows them to function effectively, and to levels of health, knowledge and education. Natural capital refers to the natural endowment of air, land, water, wildlife and the ecological services provided by healthy ecosystems (Pearce and Turner, 1990).

The theoretical basis for applying these frameworks to the measurement of sustainable development is discussed in Pearce and Turner (1990) and Atkinson et al.,

(1997). Their use requires that each form of capital stock be measured to determine how it is changing over time. However, the substitutability debate described earlier in chapter 3 is crucial here: must each type of capital stock (natural, produced, and social capital) be conserved, to which extent are capital stocks substitutable, or must only the total capital stock be preserved for sustainability?

The advantage of this framework is its comprehensiveness; conceptually it allows all factors (social, economic and environmental) and their linkages to be considered together. There are complications however - the definition of some forms of capital (e.g. social capital) remains vague, and “non-economic” forms of capital remain difficult to value in monetary terms. Example applications of this model include work by the World Bank (1997) to measure the wealth of nations, and the work of Cobb et al., (1999) to develop an indicator of “genuine” progress. The development of environmentally-adjusted national aggregates (so-called “green” GDP) represents a partial application of this framework (Hamilton and Lutz, 1996).

4.3 Input - Output (I/O) Frameworks.

Input-Output frameworks also emerged from the economics literature, including the System of National Accounts (SNA), in which flows of inputs into and outputs from an economy are articulated in money terms. Their extension to include environmental resources involves tracking environmental inputs (e.g., land, water) into the economy and outputs from the economy (usually waste residuals, such as greenhouse gases) into the environment. Linkages between resource use, commodity production, specific industrial sectors, and waste residuals can be quantified, allowing estimates of the resource and

waste intensity of an economy, or a subset of the economy, to be developed.

Daly (1968), Lontief (1970), Victor (1972) and Isard (1972) pioneered work to apply I/O models to environmental analysis, and a review of their application for such purposes can be found in Carpentier (1994), who developed an I/O economic-ecological model for Canadian agriculture. The main strength of the I/O approach is that it provides a means of identifying and quantifying the relationship between environmental assets and services and economic production, both in terms of how an economy draws upon natural resource inputs, and how waste outputs flow from the economy to the environment. As such, I/O models can reveal how changes in final demand for goods and services will affect levels of resource use and waste generation.

From a broader sustainable development perspective however, there are several drawbacks. These frameworks do not incorporate the social and human dimensions of sustainability, and the coefficients can only reflect existing technology over a short time period, thus their application to the longer time horizons associated with sustainable development considerations is limited.

4.4 Theme-Based Frameworks

Simply defined, a theme framework selects one or more themes around which an analysis of sustainable development (or some aspect of it) is performed. Examples of “themes” vary and can include an economic sector (e.g., transportation, agriculture, energy), specific environment media (soil, water, biodiversity) or even a region (e.g. Great Lakes basin, prairie region). Theme models are quite common in the sustainable development assessment literature and the use of this approach can take many shapes.

Leading examples are the World Bank's work on land quality indicators (Pieri et al., 1995) and work in The Netherlands by Adriaanse (1993). The latter presents a series of aggregate indicators for each of the following themes: climate change, acidification of the environment, disturbance of local environments, dispersion of toxic substances, disposal of solid wastes and eutrophication of the environment.

In general, theme-based assessments of the sustainability of development are not based on any consistent framework but compile a set of indicators that are felt to best reflect the various issues (themes) of interest. This flexibility is one strength of this approach. A possible drawback is the difficulty of determining linkages between the various themes, although this could be addressed in cases where a more systematic approach to link the various themes is undertaken.

4.5 Summary.

The assessment frameworks and models described in this chapter do not constitute a complete list but do provide a sense of the main frameworks used. Assessment of sustainable development is an evolving field and these frameworks originated at different stages in its development. In general, work evolved from an initial focus on specific themes (the theme-based frameworks) toward more systematic and comprehensive approaches concerned with linkages both within and, later, between systems. The stress-response frameworks and the input/output frameworks provide good examples of attempts to identify links between the environment and the economy. Such work was followed by the development of still broader frameworks which attempt to capture other aspects of sustainable development, such as the multiple capital frameworks.

However, while there are differences, there are also similarities between approaches as the models are not exclusive and can be (and often are) applied together. For example, the multiple capital approach can be seen as an extension of the theme model in that it considers the three themes of sustainable development. Similarly, it is possible to apply a variation of the stress-response approach within a theme model, with the OECD's work on the development of agri-environmental indicators providing one example (OECD, 1997a; 1999a). In all likelihood, there is a need for different approaches tailored for different applications, but a limitation inherent to the use of different approaches is the difficulty in comparing outcomes in cases where different approaches are used (Hardi et al., 1997).

5. Environmental Accounting.

This chapter provides an overview of the environmental accounting approach to assessing sustainable development. The origins of this approach are reviewed first, followed by a description of environmental accounting in its various forms.

5.1 The System of National Accounts.

Environmental accounting has its roots in the System of National Accounts (SNA), a system designed to measure economic activity developed through international collaboration initiated by the League of Nations following the Second World War. The basic purpose of the system is to record economic flows and stocks. A complete description of the SNA is available from the Inter-Secretariat Working Group on National Accounts (1993). Its complex structure is summarized below.

- “The system is built around a sequence of interconnected flow accounts linked to

different types of economic activity occurring within a specified time period, together with balance sheets that record the values of the stocks of assets and liabilities held by institutional units or sectors at the beginning and end of each accounting period. There is a direct link between the flow accounts and the balance sheets, as all of the changes occurring over time that affect the balance of assets or liabilities held by institutional sectors or units are recorded in one or another of the flow accounts.

- A series of current accounts record the production of goods and services, the generation of incomes by production, the subsequent distribution and redistribution of incomes among institutional units, and the use of incomes for purposes of either consumption or saving. Of the current accounts, a production account records the production of goods and services and is used to derive gross value-added, defined as the value of output less intermediate consumption, providing a measure of the contribution to GDP of individual producers, industries or sectors. A series of income accounts record how income is generated through production, distributed to institutional units, redistributed among institutional units (mainly by governments) and eventually used by institutional sectors for purposes of final consumption or saving.
- A series of accumulation accounts record the acquisition and disposal of non-financial assets as a result of transactions with other institutional units in the economy.
- The basic units of the system are the economic units capable of owning assets and incurring liabilities on their behalf (e.g. corporations and businesses). These are grouped together to form institutional sectors based on their main functions, the principal sectors being non-financial corporations, financial corporations, general government, households, and non-profit institutions serving households. Products (goods and services) produced by economic units are specified using the Central Product Classification” (Inter-Secretariat Working Group on National Accounts, 1993).

The SNA produces certain macro-economic aggregates that are widely used by analysts, policy-makers, the business community, the press and the general public as indicators of economic activity and welfare. Key among these are the following:

- Gross Domestic Product (GDP), a measure of the level of production of an economy

- Net Domestic Product (NDP), which is GDP minus depreciation of capital assets
- Savings, which is the difference between income and consumption.

5.2 Shortcomings of the SNA for Assessing Sustainable Development

Within the set of assessment frameworks reviewed in Chapter 4, the SNA is most closely aligned with the Capital Framework as it was designed to measure economic activity and economic wealth. From the perspective of measuring sustainable development, however, many analysts have critiqued the system.

In simple terms, the overall critique is that the SNA emphasizes a single social objective, GDP growth, to the detriment of other sustainable development goals, in particular social and environmental goals (Gilbert and Hafkamp, 1986). Scott (1956) was one of the first to reflect on the link between national wealth, natural wealth and the SNA, asking whether the increased stock of (economic) capital per head was accompanied by an increased stock of natural resources, or whether the former is merely compensating for a decline in the latter. He suggested that, as important inputs into economic production, information on changes in natural resources should receive attention in the national accounts but provided no concrete suggestions about how this might be done. Drechsler (1976) also recognized the role of environmental goods and services in relation to welfare and considered the question of whether national accounting aggregates such as GDP should be modified to reflect environmental changes. He concluded that to do so would require evaluating the monetary value of environmental changes, but that there was no sufficiently robust way of doing so, suggesting instead that the national accounting information be supplemented with bio-physical indicators of environmental quality.

As concern about environmental quality mounted in the 1970s, and about sustainable development more broadly in the 1980s, the need for environmental information and analysis grew, as did criticism of the SNA as a vehicle for informing sustainable development policy. Many among those who have critiqued the SNA draw on the concept of income, defined by Hicks (1946) as the maximum amount an individual can consume without reducing his future capacity to consume. Building on this, El Serafy and Lutz (1989) suggested GDP does not represent true, sustainable income because of two shortcomings. One concerns the treatment of so-called “defensive” expenditures, such as the cost of environmental protection activities, which the SNA treats as income. The second is that GDP does not adequately reflect environmental and natural resource degradation. Similarly, Prudham and Lonergan (1993) suggest the implication for national accounting is that any net loss of natural capital due to economic activity must be netted out of the traditional economic measures of income.

Repetto et al., (1989) stated in more popular terms that “a country could exhaust its mineral resources, cut down its forests, erode its soils, pollute its aquifers and hunt its wildlife to extinction, but measured income (i.e. GDP) would not be affected as these assets disappeared”. Harrison (1989) broadened the critique of the SNA by arguing its deficiencies are both of omission and commission. Those of omission include the concerns about the environment mentioned above, as well as failure to record such things as unpaid but crucial work undertaken by women in developing countries (such as gathering firewood and water). Cobb et al., (1999) point out similar shortcomings in the social area. For example, free services, such as community volunteer work, are not

counted, resulting in the under-estimation of social capital.

In summary, by the late 1980s, the shortcomings of the SNA in relation to the goals of sustainable development had become a cause of some concern and were receiving considerable attention from economists and others concerned with sustainable development. Some felt that if environmental and social factors could somehow be integrated into the national accounts, these factors would more likely be given due consideration in the policy development process, leading to more sustainable forms of economic development.

5.3 Approaches to Environmental Accounting.

Given widespread recognition of the shortcomings of the SNA, under the auspices of several international agencies (UNSTAT, IMF, World Bank, OECD, ECE) formal work was initiated in the mid-1980s to address how the treatment of environmental assets and services might be improved in the national accounts. Work also proceeded within some national governments and research institutions. This flurry of work, including the ideas and contributions of the principal players, is well documented in the literature, of which key sources are: Marin (1978); Harrison (1989); all papers in the volumes edited by Ahmad et. al (1989) and Franz & Stahmer (1993); Bartelmus (1987); Bartelmus et al., (1991); Hamilton (1994); Peskin and Lutz (1990); and Prudham and Lonergan (1993). No single accounting model resulted from this work; rather several approaches emerged as grouped and described in Table 3.

One result of such work was the publication of a revised version of the core SNA in 1993 (Inter-Secretariat Working Group on National Accounts, 1993). Hamilton and

Lutz (1996) and United Nations (1993) summarize the major changes from an environmental perspective:

- “The definition of what constitutes an asset was broadened through a redefinition of the production boundary. In essence, all assets that contribute to marketable production, such as land, subsoil resources, cultivated plants and livestock were included. However, it was also recognized that this change still remained inadequate from an ecological perspective. Tropical forests, for example, might have a market value through their yield of marketable wood while their other ecological functions, such as for storing carbon and as habitat, are still excluded. The revised SNA suggests such non-market functions also be recognized and valued, but the system structure was not modified for this purpose”.

Table 3: Approaches to Environmental Accounting		
Approach	Environmental categories to be taken into account	Accounting characteristics
Adjustment of SNA	Monetary valuation of: - environmental damages - environmental services - stock of natural capital - environmental expenditures	Modification of SNA framework and boundaries
Satellite accounts	Valuation of - environmental damages - environmental services - stock of natural capital - environmental expenditures Calculation of environmentally-adjusted aggregates Corresponding physical flows and stocks	Complements SNA without modifying it; retains general coherence with SNA
Natural resource and environmental accounts	Physical flows and stocks of natural resources Physical and monetary flows associated with anthropogenic exploitation of natural resources	Independent from SNA

Source: OECD, 1994.

Full integration of environmental factors into the SNA was not considered

feasible because of the limited case study work that had been done up to that time, and because of difficulties in assigning a monetary value to many environmental assets and, in particular, to non-market environmental services. Instead, it was suggested that satellite accounts be developed for the environment as a companion to the SNA. To provide guidance on how this might be done, in 1993 the United Nations published a handbook for Integrated Environmental and Economic Accounting (United Nations, 1993). This was to provide a conceptual basis for a satellite system to the SNA for integrated environmental and economic accounting (SEEA).

The 1993 SEEA includes the following:

- A basic description of production and consumption activities and of non-financial assets. The starting point for the natural asset accounts of the SEEA are the non-financial asset accounts of the SNA which also comprise non-produced natural assets.
- A description of the interrelationships between the natural environment and the economy in physical terms. This section incorporates the relevant concepts and methods of natural resource accounting, material/energy balances and input-output tabulations. Here, physical data are included on the flows of natural resources (e.g., water) from the natural environment to the economy, their transformation within the economy, and on the flows of residuals of economic activities to the natural environment (e.g., pollutants). This is accomplished through the input-output framework which can be modified to include flows of natural resources into the economy and flows of waste residuals from the

economy into the environment.

Different approaches for estimating the imputed costs of the use of natural assets were proposed. For the use of natural resources for which markets exist (e.g. minerals, timber from forests), market valuation is suggested using either the “net price” or “user cost” methods (Alfieri and Bartelmus, 1998). The net price method measures the value of the net change in the stock of the resource over the accounting period, calculated as follows:

- a) market prices of natural resource outputs are determined at the beginning, end, and average over the accounting period;
- b) total factor costs per unit of output are determined at the beginning, end and average over the accounting period;
- c) net price is determined as the difference between a and b
- d) net price is applied to opening stocks at beginning of accounting period
- e) net price is applied to closing stocks at end of accounting period
- f) environmental change is calculated as the balance between d and e (appreciation if +ve; depreciation if -ve).

The user cost method, which is considered more appropriate for non-renewable natural resources, is calculated as follows:

- a) the lifetime of the resource is estimated at current exploitation rates
- b) a discount rate is established as the opportunity cost of the next best investment
- c) net returns from sales of the resource over the accounting period are compiled
- d) a user cost allowance is calculated by applying a and b to c.

Different valuation methods are suggested for environmental services:

maintenance costs and contingent valuation. Maintenance cost uses an estimate of the costs required to sustain at least the present level of natural assets (i.e. the cost of avoiding damage). Contingent valuation assigns a value to the environment based on what people are willing to pay for a specified level of environmental quality, or their willingness to accept compensation for some degree of loss, and could be applied especially for estimating the value of the consumptive services of the natural environment.

Valuation is perhaps the most controversial aspect of environmental accounting, as different valuation methods and discount rates may produce quite different results, with key implications for benefit-cost studies and sustainable development policy decisions. Some of the methods proposed have been critiqued and alternatives have been suggested. Hamilton and Ward (1998), for example, suggest that for pollution, the marginal social (i.e. damage) cost of pollution, as measured through abatement expenditures, or through the value of welfare loss, are more appropriate measures than the maintenance cost of pollution prevention. Other analysts have suggested alternative approaches. Heuting (1989), and Heuting and Bosch (1990), for example, advocate a variation of the maintenance cost approach based on the cost of achieving a level of environmental quality set by policy or through technical standards. The economic valuation issue is discussed further in chapter 7 of this report.

The different forms of environmental accounting listed in Table 3 have alternatively been referred to as natural resource accounting, “green” accounting,

environmental accounting and integrated economic-environmental accounting. Natural resource and environmental accounting aims to collect information on stocks of natural resources and flows of resources between the environment and the economy, expressed primarily in physical terms. In satellite accounting (the SEEA), a formal structure is proposed which builds on and makes use of the structure of the SNA itself, including balance sheets and input-output matrices, allowing analysis of changes in environmental stocks, and resource use and waste generation by commodity, industry or sector, and unit of final demand, expressed in both physical and monetary units. The final method identified, environmental adjustments to the SNA, are limited to those which can be made within the system without compromising its structure and continuity. In practice, this means that only those environmental stocks and flows for which market-based monetary values can be derived are included, such as mineral reserves.

There are obvious links between these versions as all are based on the core concept of stocks and flows to, within and from the economy. There are also important differences. Natural resource accounting is the most basic form of environmental accounting. It provides the initial physical measures of environmental stocks and flows and derives more of its foundations from the ecological sciences than do the other two approaches, thus it is more closely aligned with the so-called “strong” view of sustainability (Pearce et al., 1996). It also provides a base from which the other forms of accounting can be built. The two other approaches move beyond a biophysical approach to also include monetary valuation of changes in environmental stocks and flows, and the calculation of environmentally adjusted aggregate indicators such as Environmentally

Adjusted Net Domestic Product (Bartelmus et al., 1991) or “genuine savings” (World Bank, 1997). Because such aggregate measures implicitly allow for the substitution of different forms of capital, they are more closely aligned with a “weak” view of sustainability (Pearce et al., 1996).

6. Example Applications of Environmental Accounting.

Over the past 15 years, much theoretical and operational work has been undertaken to develop environmental accounts for specific areas and issues. In this section, examples of environmental accounting applications are presented for land, agriculture, and forestry. These were selected to illustrate both biophysical and economic approaches to environmental accounting in the natural resource sectors. In cases where economic income is adjusted for environmental costs and benefits, the monetary valuation methods used are reviewed. Readers interested in more comprehensive review of environmental accounting applications (including in other sectors) are directed to OECD (1994), Hamilton and Lutz (1996), and Uno and Bartelmus (1998).

6.1 Land Accounts.

Three examples of land accounts are reviewed here: a study by Radermacher (1998) in Germany, a land account developed by Statistics Canada (1997a, 1997b), and a study by Adger and Whitby (1993) of the land-use sector in the UK.

6.1.1 Land accounting in Germany. Radermacher’s study of land use accounting in Germany deals in biophysical units only. In the study, a set of core and supplementary accounts are developed, each composed of three levels: stocks, total flow/change of stocks during the accounting period, and the disaggregation of flows. To facilitate use of

existing data, land use classifications were selected or derived from existing European or German land cover systems. Illustrative examples of the approach are provided below for land use and land cover.

Table 4: Simplified matrix of land use and land cover stocks							
Land Use	Land Cover					Land Use Total	Land Use (%)
	Artificial cover	Woody Vegetation	Grass Cover	Water surfaces	Other		
Dwellings	874	3421	11716	---	---	15141	61.2
Recreation areas	79						
Agriculture	---						
Forestry	3024						
Other	2233						
land cover							
Total land cover	9.0						
(%)							

Source: Radermacher, 1998.

Table 4 identifies how land stocks are distributed by use. Numbers are entered here for only agriculture and artificial cover (a completed matrix is provided in the original paper) showing how the total land stock for each is distributed by use (in the case of artificial cover) and cover (in the case of agriculture). A separate matrix (not shown) is developed to track flows between land use types, such as between agriculture and built-up areas, recreation areas, forest areas and other uses. Finally, a summary balance sheet of land by use class is developed showing opening and closing stocks, and net flows, over the accounting period (Table 5). In addition to the above core accounts, more detailed supplementary accounts are discussed. Although not shown in the original paper, one of

these describes the demand for land by different economic sectors. The other, intended to describe the environmental impact of this demand, was not developed in the study.

Radermacher concludes that an accounting system for analysis of land use offers several advantages: highly aggregated information can be produced on different environmental issues, relationships between different environmental issues can be analyzed and determined, aggregate indicators can be derived from the accounts, and an accounting approach offers opportunities for checking data consistency. He also discusses the potential link between environmental accounting and the Pressure-State-Response framework approach, suggesting that a sectoral breakdown of land use activities can provide a first approximation for specific pressure indicators (such as artificiality of land cover as a measure of human impact on land, vegetation, etc). He does caution, however, that accounting is limited in its ability to identify direct cause-effect linkages, and that even after the study, the statistical linkage of the economic activities with accounts on the state of the environment is still unsatisfactory. It is also clear that while the accounts provide useful information on changes in land use, no information is provided on the significance of these changes from a sustainable development perspective; this is left open to interpretation by users of this information.

Table 5: Core account, land use stocks 1972/92 - flows, hectares				
Land use	Opening stock (1972)	Net flow		Closing stock (1992)
		Due to human activity	Due to natural processes	
Built-up land	3,893.8	1,128.7	- 4.7	5,017.8
Quarries, pits, mines, dumps	305.9	- 81.1	- 107.3	116.7
Transportation	456.7	- 80.2	0.0	376.7
Recreation	289.6	204.5	- 2.8	491.2
Agriculture	12,363.0	- 1,025.0	- 80.4	11,257.7
Forest	7,041.8	31.2	33.2	7,105.3
Water	171.3	32.9	0.0	203.9
Wetlands, arid land, sparse vegetation	0.0	0.0	89.0	89.0
Fallow land	228.3	-210.2	73.1	91.2
Total area	24,750.5	4136.1	221.1	24,750.5

Source: Radermacher, 1998.

6.1.2 Statistics' Canada's land account. Statistics Canada operates a formal

environmental accounting program to describe and measure relationships between the environment and the economy (Statistics Canada, 1997a). This system contains three main components, as follows.

- "A Natural Resource Stock Account (NRSA), which measures quantities of natural resource stocks and the changes to these due to natural and human factors. These stocks are recorded in both physical and monetary units and provide a basis for estimating Canada's natural resource wealth. The NSRA considers stocks of subsoil resources (minerals, oil, etc), timber and land.
- A Material and Energy Flow Account, which records, in physical terms only at this stage, the flows of materials and energy (in the form of natural resources and wastes) between the environment and the economy. This account is linked directly with the Input-Output Accounts and allows for calculation of the resource and waste intensiveness of the economy over time. Flows are related to the activities

of industries, households and governments, providing a measure of the economy's requirement for natural resources and its use of the environment as a sink for wastes. Trends over time can be used to develop indicators of resource use efficiency. Flows are measured in biophysical units; monetary valuation is not used at this time due to the difficulty of assigning damage costs to emissions of wastes into the environment.

- An Environmental Protection Expenditure Account, which identifies current and capital expenditures by businesses, governments and households for environmental protection. The account provides a measure of financial outlays associated with environmental conservation and protection, as well as the contribution of environmental management to overall economic activity”(Statistics Canada, 1997a).

The Land Account is part of the NRSA and is intended to track the distribution, use and quality of land, the monetary value of land, and how these factors are changing over time. The account includes land cover information (vegetation, ice, water, built-up land, and other features), land use in agriculture, forestry, urban and rural use, land potential for various uses as defined in the Canada Land Inventory, and land value.

The agriculture portion of the Land Account is derived primarily from the data and classification systems used in Census of Agriculture and the Canada Land Inventory (CLI) (Environment Canada, 1978). Detailed data describe amount and location of land by CLI Land Capability Class, and also the location and amount of land under different crops, pasture, summerfallow and other agricultural land. Information can also be aggregated as either improved or unimproved farmland.

An indicator of agricultural land use and supply developed from the agricultural land account (Figure 2) provides an example of how this account has been used to assess sustainability in agriculture. Information on land under cultivation (cropland and summerfallow) has been combined with information on prime farmland availability

Figure 2: Agricultural land under cultivation relative to the stock of dependable land, 1901-1996 (Statistics Canada, 1997b).

(defined as CLI classes 1-3). The indicator shows that land under cultivation now exceeds the supply of dependable farmland, with potential implications to soil productivity.

The Land Account for agriculture also includes a valuation component. The valuation method used, reported farmland market values, is consistent with the valuation method used in the System of National Accounts where valuation is based on prevailing market prices. However, Statistics Canada recognizes a deficiency with this method of valuation, as the market value of agricultural land may be heavily influenced by speculative markets. As such, this methodology likely overestimates the agricultural value of some land and may be more reflective of its value in uses other than agriculture, such as urban development. The contribution of agricultural land value to national wealth is included in a separate indicator of natural resource assets and national wealth (Statistics Canada, 1997b).

As with the study by Radermacher (1998), Statistics Canada's land account demonstrates how environmental accounting can be applied to land-based sectors. A practical approach is taken where the accounting classifications used are derived from those already developed in existing databases. The agriculture application, however, is a quantitative accounting of the spatial extent of land use which neglects aspects of soil quality. Valuation does not account for the effects of changes in quality either on the farm (e.g., possibly reduced yields or higher fertilizer costs) or off the farm (e.g., reduced water quality from sedimentation, excessive nutrients or pesticide loadings), which would have required the use of other methods, such as some of those proposed for the SEEA.

6.1.3 Land accounting in the U.K. In a study by Adger and Whitby (1993), an accounting

approach was used to assess the role of the rural land use sector (defined as agricultural, forestry and recreational use of land) in the UK economy. Income from the sector is adjusted for three environmental effects: emissions and sinks of carbon, nitrogen pollution from agriculture, and non-market service flows from recreational lands. These adjustments are summarized as follows:

$$\text{Sustainable Net Income} = \text{Net Product} - C_D - K_N, \text{ where}$$

$$\text{Net Product} = \text{gross income from the sector less } K_r \text{ (change in reproducible capital stock)}$$

$$C_D = \text{defensive expenditure or consumption by households and governments}$$

$$K_N = \text{change in natural capital stock}$$

The modified welfare measure is:

$$\text{Modified Net Welfare} = \text{Sustainable Net Income} + \text{NMS}$$

where NMS = annual non-marketed service flows from the land use sector.

Since this analysis seeks to adjust income, it draws on the approaches and methods proposed for satellite environmental accounting in the SEEA, including several of the proposed economic valuation methods for valuing environmental effects (costs and benefits). For carbon emissions, a carbon budget was constructed for the land sector which revealed a net fixation in 1988 of 4.3 million tonnes of carbon. A social cost estimate of 30.7 pounds sterling / tonne C was determined using a maintenance cost approach which considered the costs of switching to alternative, non-carbon-emitting energy sources that would reduce carbon by this amount in the forestry sector given present and projected technologies.

For nitrogen, a budget was constructed showing additions, stocks and losses of nitrogen. Valuation was restricted to the costs of water pollution, estimated as the benefit

of reduced agricultural nitrate pollution per household surveyed derived in a previous study (13 pounds per household, equivalent to 10.8 million pounds nationally). Defensive expenditures were calculated from government expenditures on national parks, payments to maintain other special sites of significance and the costs of river cleanup due to pollution (total of 58 million pounds). These expenditures were subtracted from gross income statistics. Finally, the value of positive externalities of the land sector, in the form of landscape, wildlife and recreational benefits, was obtained from a series of contingent valuation studies. In total, a positive service flow of some 888 million pounds was estimated for such benefits. The overall results of the study are presented in Table 6.

Table 6: Income adjustments from environmental accounting in the UK land sector	
Factors	Value (1988 pounds)
Net domestic product from land sector	+ 4028
Environmental adjustments	
- net carbon sink	+ 132
- nitrate pollution	- 11
- defensive expenditures	- 58
- non-market service flows	+ 888
Environmentally-adjusted NDP	+ 4979

Source: Adger & Whitby, 1993.

Adger and Whitby's work represents a multi-sector application of environmental accounting in the land sector. The study shows how both positive and negative externalities can be taken into account to adjust gross income, and illustrates the use of several valuation techniques which are broadly consistent with the valuation techniques identified for the SEEA. The authors do caution, however, that the usefulness of the

adjustment made toward a net welfare measure depends on the acceptance of the assumptions involved in the valuation studies. They also suggest that other adjustments would be needed to provide a more complete picture of how land-based industries relate to the economy and the environment. It is also clear that this analysis corresponds to a weak sustainability perspective. Net domestic product is adjusted upward, largely on the strength of the value people attach to landscape, which masks the environmental degradation from nitrate pollution.

6.2 Environmental Accounting in Agriculture.

The approaches used in six separate studies are reviewed here to illustrate how environmental accounting has been applied in the agriculture sector. Work by Repetto et al., (1989), Repetto (1992), Solorzano et al., (1991), Hrubovcak et al., (1995) and Faeth et al., (1991) deal with the on-farm costs of soil erosion. Hrubovcak et al., (1995) and Faeth et al., (1991) also deal with the off-farm costs of erosion, and work by Malik (1994) illustrates an application of environmental accounting methodology to groundwater.

6.2.1 Studies of Indonesia and Costa Rica. In studies of agriculture (and other resource sectors) in Indonesia and Costa Rica, Repetto et al., (1989) and Repetto (1992) consider the costs of soil erosion using two different valuation methods. All three studies first develop a physical land account in which both land use and soil erosion are estimated.

Repetto et al., (1989) estimate the cost of erosion using a crop productivity approach. Yield-erosion relationships were obtained for the two main crop groups and various soil types. Impacts on yield were calculated based on estimated erosion rates and erosion-productivity relationships. Representative farm budgets were used to model

possible adjustments by farmers (for e.g., increased use of fertilizer) and thus overall impacts on farm income from erosion and its related effects. The single-year (1985 in this study) cost estimates of erosion were then capitalized to obtain the total present value of the future stream of productivity losses associated with the erosion estimated for that year, which is taken as the economic measure of soil depreciation. Selected study results are presented in Table 7.

Table 7: Value of crop production in Java compared with the cost of erosion (million rupiah), 1985	
Value of crop output (6 major crops)	1,344,356
Cost of single year erosion loss	53,956
Capitalized value of erosion losses	539,560
Single year erosion cost as fraction of value of output	0.04

Source: Repetto et al., 1989.

The study concluded that nearly 40 cents in future income was being sacrificed to obtain each dollar for current consumption. Costs of resource use for oil and timber were also estimated in this study using the net price valuation method. The study adjusts national GDP for resource depletion and presents the results as net domestic product (these results are not presented here).

Repetto (1992) estimates the change in the value of Costa Rica's natural resources over the 1970-1989 period. Opening and closing stocks of land resources and soil losses from erosion were calculated for each year. The costs of erosion were estimated as the cost of replacing lost nutrients with fertilizer nutrients; other environmental costs of erosion such as compaction were not assessed. Still, results showed that the cost of

erosion using this method was equal to a significant proportion of the total value of agricultural production (9% in 1984 for example). A downward adjustment of about 7% of agricultural GDP was estimated in the last four years of the study.

Both studies use similar methods for estimating physical erosion rates, but different approaches to valuation. Unfortunately, it is difficult to compare the valuation results because the studies value total agricultural output differently.

6.2.2 Accounting for the environment in US agriculture - USDA study. In a more recent study, economists in the US Department of Agriculture (Hrubovcak et al., 1995) attempted to more accurately measure agriculture's contribution to national income. The point of departure for estimating agricultural income is agriculture's share of Net National Product (NNP, which is GDP adjusted for depreciation of produced assets). Agricultural NNP is then further adjusted for environmental and resource changes stemming from soil erosion, depletion of groundwater and off-farm water quality impacts. This study therefore incorporates a wider range of impacts than the studies described earlier.

Several methods are used to value environmental damage in monetary terms. Farm productivity loss estimates due to soil erosion were taken from another USDA study (USDA, 1989). These represent, in aggregate, a 3% loss in productivity over the next 100 years assuming farm and management practices remained unchanged from 1982 (estimated productivity impacts ranged from a high of 7.1% in the Northeast region to negligible in some other regions). Productivity decreases were then linked with crop prices to develop an estimate of economic losses attributable to soil erosion over time.

The impacts of erosion on off-site water quality were obtained from another study (Ribaudó, 1989): total damage costs from erosion of \$1.78 US (1986 dollars) per ton of soil eroded or \$9.00 per acre of land were estimated, with a range of between \$0.57 - \$7.06 per tonne of soil eroded, depending on the region. These estimates consider the damages on freshwater and marine recreation, water storage, navigation, flooding, roadside ditches, irrigation ditches, freshwater and marine commercial fishing and other uses. The value of total damages was calculated by applying Ribaudó's per-ton estimates to the total level of erosion.

A variety of methods were used to estimate the off-site erosion damage costs, following into three categories: changes in defensive expenditures, changes in production costs, or changes in consumer surplus. An example of how each method was used is provided below.

- **“Defensive expenditures method:** estimates of defensive expenditures for weed control and ditch cleaning were obtained by region from available studies and government agencies (for example, data on ditch maintenance costs were obtained from State highway departments). A linear relationship between annual costs and levels of erosion was assumed in which a percentage change in erosion results in the same percentage change in maintenance costs.
- **Production cost method:** used to estimate costs of water treatment due to erosion. A previously-developed water treatment cost model was used to specify a water plant's treatment costs as a function of water production, influent water quality (turbidity), distribution costs and the prices of other inputs. The model was then used to estimate the cost of reducing one unit of turbidity in one gallon of water. Other equations were used to relate changes in soil erosion to changes in total suspended solid concentrations in water, and the estimated change in turbidity was then related to the per-unit treatment cost to arrive at an estimate of water treatment costs due to soil erosion.
- **The change in consumer surplus approach** was used to estimate costs to freshwater and marine recreation. A model was used that links changes in water

quality to changes in recreational activity. This model estimates the proportion of people who will fish in a given year, and how often. It was assumed that both decisions are a function of changes in water quality. A dollar value for changes in fishing days was obtained from other sources and the costs to freshwater and marine recreation of declines in water from erosion were thus produced” (Ribaud, 1989).

Effects on groundwater stocks over time were also considered. First, national and regional data were used to determine changes in groundwater stocks (for those regions where such stocks are in net decline) and agriculture’s share in total withdrawals. These data were then used to estimate agriculture’s share in the rate of depletion of groundwater. A final step required estimating the value of groundwater, which was done using the value of energy expenses for on-farm pumping of irrigation water from wells. Data on energy expenses were obtained from other federal sources, with values ranging from \$0.10 to \$0.12 per 1000 gallons to \$0.07 to \$0.09 per 1000 gallons. The adjustment combines the value of ground water with the rate of groundwater depletion associated with agriculture. Selected results of this study are presented in Table 8.

Table 8: Adjusted US Farm Income, 1982, 1987 and 1992 (billion US dollars)			
Income Component	1982	1987	1992
Traditional farm income	54.0	54.3	66.8
Water quality			
- Industry transfer	- 2.4	- 2.7	- 2.7
- Consumer effects	- 1.1	- 1.2	- 1.3
Water quantity	- 0.2	- 0.2	- 0.3
Adjusted (green) farm income	50.2	50.2	62.5

Source: Hrubovcak et al., 1995.

In each year of this study, agricultural income is reduced by about \$4 billion after

accounting for agriculture's contribution to declines in surface water quality and groundwater quantity. The study concludes that environmental problems in US agriculture exist and the resource base is depreciating, but the extent of the effects is within the range that can be adequately addressed by careful policy. The authors also mention that the estimates of "green" or adjusted income are incomplete, and that other adjustments, including additional environmental damages and valuation of environmental services, are required to develop a credible measure of agricultural sustainability.

6.2.3 Accounting for the environment in US agriculture - WRI study. In another study of US agriculture by the World Resource Institute (WRI), Faeth et al., (1991) apply an environmental accounting approach for assessing the "real" costs of US agricultural commodity support, and adjusting farm income to reflect such costs. Through two case studies, one in Nebraska and the other in Pennsylvania, farm income is compared under commonly used conventional farming systems and alternative systems. Conventional systems are defined as those which rely on heavy inputs of fertilizers and pesticides, whereas alternative systems rely on crop rotations and tillage practices for soil fertility and moisture and pest management.

The study simulates soil erosion rates (tons/acre/yr) and the economic value of off-farm erosion impacts and on-farm soil depreciation under various tillage and crop rotation combinations in the two regions. Erosion rates and associated on-farm productivity effects were estimated with the Erosion-Productivity Impact Calculator (EPIC) model. The present-value of economic losses from reduced yields was calculated over a 30 year period using a 5 percent discount rate. As in the study by Hrubovcak et al.,

(1995), the off-site erosion cost estimates developed by Ribaudo (1989) were used to estimate water quality impacts from soil losses. The regional damage cost factor relevant to each study region was selected and updated using a multiplier (\$8.16/ton/yr for Pennsylvania; \$0.66/ton/yr for Nebraska).

Table 9 presents selected results of this study for Pennsylvania. Net farm income is adjusted for the cost of soil depreciation and the off-site costs of sedimentation from erosion. In this study, an \$80.00/acre/year profit as calculated by conventional accounting (\$45.00 + \$35.00 in commodity subsidies) is reduced to \$8.00/acre/year under environmental accounting procedures which consider on- and off-farm erosion costs.

Table 9: Net Farm Income Calculated Using Conventional and Environmental Accounting Methodology (\$/acre/year)		
Factor	Conventional Accounting	Environmental Accounting
Gross operating margin	45	45
plus commodity subsidies	35	35
minus soil depreciation	–	- 25
minus off-site costs	–	- 47
Net farm income	80	8

Source: Faeth et al., 1991.

6.2.4 Accounting for water use in Indian agriculture. Malik (1994) considers the economic cost over a twenty-year period of agricultural use of groundwater in the Indian states of Punjab and Haryana under two farming systems: rice-wheat and maize-wheat, and then adjusts farm income to account for environmental costs. The study is thus conceptually similar to the study by Hrubovcak et al. (1995) but takes fewer

environmental cost factors into account.

A simple water account is constructed from available data showing the prevailing depth to the water table and its estimated rate of decline over the study period. The value of the water resource is estimated as the likely increase in the farmer's cost of pumping water due to a decline in the water table over the study period (similar to the valuation method used by Hrubovcak et al., (1995) for this resource). Farmers were surveyed to determine water pumping behaviours (e.g., frequency of changes in pumping equipment due to declines in the water table) and capital costs of pumping equipment were obtained at market prices. On the basis of such information the likely changes in capital and operating costs of pumping equipment were estimated for a representative two-hectare farm, and the value of groundwater per hectare was estimated using three discount rates. Projected increases in the extraction cost for each period are discounted and summed to obtain the present value of the total resource cost of groundwater depletion. Results of this study are presented in Table 10.

The results of this study show the implications of including and excluding the cost of environmental degradation from estimates of farm income. The rice-wheat farming system is more demanding of groundwater resources than the maize-wheat system, but farm income as traditionally measured is 54% greater from rice-wheat cultivation than for maize-wheat. However, if the costs of groundwater depletion are considered and deducted from income, the difference in net financial values decreases from 54% to between 27% and 30%, depending on the choice of discount rate. Clearly, economic analysis that excludes the value of resource-degrading activities overestimated the value of such

activities. The study results nonetheless suggest the more water intensive farming system provides greater net benefits to society (if only water use costs are considered). An interpretation based on strong sustainability or more ecological principles could, however, lead to a different conclusion.

Table 10: Farm income with and without adjustment for environmental costs									
Income (`000 rupees)	Farming system						Percent difference		
	Rice-wheat			Maize-wheat					
	Discount rate (%)								
	6 (2)	8 (3)	10 (4)	6 (5)	8 (6)	10 (7)	6 (8)	8 (9)	10 (10)
Un- adjusted returns	1.58	1.36	1.17	1.03	.88	.76	54	54	54
Cost of ground- water use	.28	.23	.18	0	0	0			
Net returns	1.3	1.13	.99	1.03	.88	.76	27	29	30

Source: Malik, 1994

6.3 Environmental Accounting in Forestry

Environmental accounting has been widely applied to analysis of forest resources. Examples of studies include Koltola (1994), Solorzano et al. (1991), Anielski (1992), Repetto et al. (1989), Hultkrantz (1992) and Liu (1998). Of these, Liu's study of forests in China is profiled here as it offers a recent example of how environmental accounting has been applied in this sector.

The starting point is the development of a biophysical forest account for China as a whole. Inadequacies in available statistics were addressed using different methods and assumptions. For example, timber stocks refer to all standing trees and not usable timber as no data were available to sub-classify by age, height and diameter. The basic account thus considers, for each year from 1976 through 1992:

- 1) opening stocks (million M³)
- 2) depletion due to each of consumption, withering, fire damage, and insect damage
- 3) additions from natural growth
- 4) net change, calculated as (1) minus (2) plus (3) (always negative in this study).

Two methods are used to determine the economic value of the net depreciation in forest resources: net price and user cost. The first step is calculation of stumpage value, defined as the value of stocks based on timber sale proceeds less the costs of logging, transportation and processing. A single stumpage value is derived for the whole economy for each year, based on international prices and cost data obtained from a specific forest and extrapolated to all forest areas of the country. Using the net price approach, the economic value of the depreciation of timber is obtained by multiplying each year's volume loss of timber by the corresponding average unit stumpage value.

The second valuation approach, user cost (El Serafy, 1989) requires determining the amount that should be set aside to keep income-generating investments in the future. User cost was calculated residually from estimates of other factors: value added (gross output value - intermediate cost of production) minus capital remuneration minus wages calculated over a 50 to 100 year period. Discount rates of 0%, 5% and 10% were used but

differences were found to be small. Summary results are presented in Table 11.

Differences in adjusted forestry GDP illustrate how different depreciation methods can yield different results. The study's author recognizes but does not explain this difference. Both valuation approaches reflect the real scarcity of forest resources, and both, in combination with the biophysical account, show a depreciation of forests in China due to human activity. The author illustrates how environmental accounting can help inform policy by concluding that forest policy reform is needed. He suggests such a policy for China would consist of a timber import strategy for at least the next decade, timber substitution and savings policies, and more investment in reforestation and plantation.

Table 11: Adjusted gross forest product for China using two depreciation approaches (million yuan, 1980 prices)					
Year	Real GDP	Depreciation (user cost)	Adjusted GDP (user cost)	Depreciation (net price)	Adjusted GDP (net price)
1980	175.6	107.4	68.2	22.8	152.8
1981	156.2	74.5	81.7	33.3	122.9
1982	148.9	64.0	84.9	51.8	97.1
1983	131.4	44.9	86.5	49.8	81.6
1984	190.9	101.2	89.7	46.4	144.5
1985	189.5	104.5	85.0	47.6	141.9
1986	227.7	137.9	89.8	52.7	175.0
1987	186.1	99.9	86.2	95.7	90.4
1988	161.0	62.8	98.2	50.2	110.8
1989	150.7	59.5	91.2	83.1	67.6
1990	220.8	134.8	86.0	58.4	162.4
1991	192.5	107.7	84.8	60.7	131.8
1992	240.0	148.5	91.5	57.0	183.0

Source: Liu, 1998.

6.4 Summary.

The examples of environmental accounting presented here provide concrete illustrations of how this approach to assessing sustainable development has been applied to the natural resource sectors of land, agriculture, and forestry. Initial accounting in physical units is common to all the studies, but beyond that there are important differences of approach. These differences relate to which factors are taken into account and how these are valued. Naturally, these factors lead, in turn, to differences in how results are interpreted.

The studies by Radermacher, Adger and Whitby, and Statistics Canada together illustrate how stocks and flows of land can be measured, and how the land resource interacts with other environmental media and various economic sectors. The studies that apply environmental accounting to the agriculture sector differ in the range of environmental factors considered. All, save for the study by Malik (1994), consider the effects of erosion on land productivity and farm income using broadly similar methods, and most also consider the impacts of agriculture on other environmental resources off the farm. There are differences, however, in which factors are considered and also, to some extent, in how the environmental costs of agriculture off-site are valued in monetary terms. The study by Liu shows very clearly how different valuation methods can yield different results.

The studies suggest the linking of environmental data with economic data, including to specific economic sectors, can yield valuable insights into where environmental improvements within and across economic sectors are most needed.

However, all study authors urge caution regarding interpretation and use of their results, pointing out that not all factors are taken into account, and that many would question some of the assumptions used in the valuation methodologies, and thus the interpretation of the results. Hrubovcak et al., (1995) for example, suggest the net economic benefits of agriculture production far outweigh the costs, and Adger & Whitby (1993) show the positive externalities from land in the UK to exceed the negative ones. Faeth et al., (1991), on the other hand, revise farm income down to near-negative margins when on-site and off-site erosion effects are considered.

A key observation concerns how results are interpreted. As mentioned above, no study claims to take all factors into account, but those that adjust income to account for environmental change (both positive and negative) show a net benefit from the economic activity considered. However, an interpretation of results based on a more ecocentric, ecological or stronger view of sustainability would likely argue that any environmental cost resulting from economic activity denotes unsustainability. This, in turn, raises the matter of the significance of the changes measured, which all reviewed studies are largely silent on.

Perhaps the overall conclusion to be reached is that the examples reviewed here show that no single and uniform approach to environmental accounting has yet emerged, even when applied within specific sectors. However, this flexibility may well be a strength of the accounting approach, and all examples share certain commonalities, such as consideration of stocks, material exchanges and flows, and quantification of changes and impacts in physical and/or economic units. Consideration of the findings and

conclusions reached in these studies suggests environmental accounting can provide valuable insights into the environmental and economic benefits and costs of economic activity. The interpretation of results, however, is very much a function of one's subjective world view, which neither physical or monetary measures can adequately weigh in a manner that would likely reflect the various views of sustainability held across society.

7. Outstanding Issues in Environmental Accounting.

Theoretical and practical work has advanced the field of environmental accounting considerably over the past 15 years. However, just as environmental accounting was initiated in response to the criticisms and shortcomings of the SNA, the application of this methodology has itself been criticized and remains controversial in several respects. The key points of debate revolve around monetary valuation of the environment (whether to and if so, how?) and the substitutability of natural capital for other forms of capital. Also, technical issues remain that have yet to be fully resolved. An overview of such issues is presented here.

Environmental accounting draws on both the economic and natural sciences, and differences in these paradigms are at the root of much debate. Spash (1997), for example, observes that economics is primarily concerned with tradeoffs at the margin whereas the natural science approach is concerned primarily with limits. These two approaches converge in the discipline of ecological economics, which, he claims, has yet to reconcile these views and thus remains methodologically disunited.

The natural science view as presented by Wackernagel (1999), Rees and

Wackernagel (1999), Ropke (1999), Herendeen (1999), and Vatn and Bromley (1994) lend support to Spash's observation. Wackernagel (1999) considers what minimal principles should be applied as criteria to ensure that ecological economics studies are consistent with the recognition of the earth's carrying capacity. His first proposition in that regard is that ecological economics studies must be grounded in biophysical assessments and compatible with scientific evidence pertaining to the integrity of ecosystems. He extends this proposition by suggesting that monetary evaluations should no longer be considered in ecological economics, arguing that because such valuation considers tradeoffs of natural and produced capital at the margin, it cannot recognize the absolute limits of substitutability whereas biophysical assessments can. Rees and Wackernagel (1999) advance similar arguments, stating that "marginal prices for nature's services and resource commodities - regardless of how these prices are assigned - generally do not (indeed cannot) reflect biophysical scarcity, increasing marginal risk with scarcity, factor complementarity, structural or functional necessity, unaccounted service flows, or informed social preferences. Ropke (1999) agrees that decisions made at the margin using monetary expressions of relative costs and benefits cannot take into account non-marginal decisions, whereas Herendeen (1999) suggests biophysical assessments are better able to consider critical matters of scale and limits as related to environmental impacts from economic activity. Vatn and Bromley (1994) object to economic valuation on the grounds that environmental goods and services embody characteristics that present serious complications when collective choices are to be made on the basis of recommendations derived from the aggregation of individual values

elicited by contingent valuation techniques. They argue that the resulting compression of the complex and interrelated attributes of the environment into a simple metric of money values results in an important loss of information that is not randomly distributed and thus twists the information about individual and collective preferences.

Proponents of a more economic approach have, of course, a different view. They argue that policy decisions are made at the margin, where information on relative benefits and costs is essential. Failure to value and weight these in comparable terms implicitly assigns an equal weight to all factors, and risks leading to sub-optimal decisions and outcomes. Environmental accounting is seen as a way of integrating environmental benefits and costs into economic analyses at the macro level, and such measures as green GDP are seen as providing a truer measure of societal welfare.

A variety of techniques have been developed to determine value for non-market (i.e. environmental) goods and services, of which an overview is provided in Pearce and Turner (1990) (also see sections 5.3, 6.1.3, 6.2 and 6.3 of this report). Such techniques are not without controversy, however, as several authors have pointed out that different valuation techniques will yield different estimates of the same phenomena. Aaheim and Nyborg (1995), for example, cite several cases where, for pollution, avoidance costs, repair costs, willingness to pay and willingness to accept will not be equal. Jaeger (1995) also points to potentially serious drawbacks in using willingness to pay aggregates as measures of social benefits or costs.

Another debate about environmental accounting concerns its utility in measuring sustainable development, or even environmentally-sustainable development, because

environmentally-adjusted aggregates correspond to a very weak view of sustainability. El Serafy (1997) for example, sees the SNA primarily as an economic framework and thus incapable of capturing all environmental change, further arguing that aggregate measures such as environmentally-adjusted GDP can only ensure aggregate income sustainability (or weak sustainability). Lintott (1996) expresses a similar view, arguing that environmentally-adjusted national income effectively treats investment in produced capital as a substitute for keeping natural capital intact, and output growth as a substitute for reducing environmental costs. Lintott concludes that a more appropriate approach from the perspective of achieving strong sustainability is the development of a set of social and environmental indicators separate from the national accounts, a conclusion that Dreschler (1976) had reached before serious work on environmental accounting had even begun. Pearce et al., (1996) discuss the development of indicators from the perspectives of both strong and weak sustainability, and agree that environmentally-adjusted income measures correspond more closely to a weak sustainability perspective.

Peskin (1998) reviewed environmental accounting in its various forms and observed that no single approach dominates the others. Aaheim and Nyborg (1995) observed that the purpose of environmental accounting, and in particular of environmentally-adjusted aggregates, is not clear. They note for example, that Heuting et al., (1992) see the goal of adjusted GDP as providing a measure of sustainable development, Maler (1991) sees it as a measure that corresponds better to our perception of welfare, and Peskin et al., (1992) seek a proper measure of value-added in the economy. They conclude that in the ongoing debate on this issue more attention to

defining which question a corrected national product is really meant to answer is needed.

8. Conclusion.

Assessment of sustainable development remains a very active field of research, but a universally-accepted definition and method of measurement remains elusive. All of the frameworks and methods reviewed here are in wide use, sometimes in combination, and work on their refinement is ongoing. All also suffer from various limitations and drawbacks.

Environmental accounting represents one still-evolving analytical approach to have recently come into use. There is apparent agreement among its proponents on accounting in physical terms as a core step. The advantages of environmental accounting are seen as the provision of structure, consistency of definitions and classifications, and a useful underlying framework through which environment-economy interactions can be analyzed. The accounting approach to analysis of sustainable development is being pursued in several countries and research institutions, and a considerable literature has developed in this area. However, all of the accounting approaches reviewed here remain incomplete: in some cases key environmental factors remain to be captured in the analyses, and where these are captured, particularly in monetary terms, the results are often controversial.

Monetary valuation of environmental goods and services is thus the most controversial aspect of environmental accounting. Some reject such valuation entirely whereas others argue just as strongly that this is a necessary step towards better decisions about sustainable development. Proponents of the latter view continue to work toward

improving valuation methods, particularly for the treatment of environmental externalities (Bartelmus, 1998). Work also continues under the auspices of the so-called “London Group”, a group of environmental accounting specialists recently given the task by the United Nations Statistical Office of revising the 1993 SESA manual. Among the goals of this effort are the simplification of the SESA approach and a reduced emphasis on monetary accounting, pending further clarification on appropriate valuation methodology. Publication of the revised methodology may thus provide further impetus for use of this approach to assessing sustainable development (R. Smith, personal comm., 2000).

Part C: An Environmental Accounting Framework for Examining the Relationship of Agricultural Land to Economic and Ecological Welfare.

This section develops a framework for assessing the sustainability of agricultural land in relation to economic and ecological welfare. The framework is developed here at a conceptual level and is intended to apply to Canadian agricultural conditions. Where data are included, this is done for illustrative purposes only. Linkages between agriculture and the environment are first reviewed in general terms, followed by a discussion of ways in which agricultural lands can contribute to welfare. Accounts for each major function of agricultural land are then presented. Part D of this report develops a quantitative application of a component of the framework introduced here.

9. Agriculture and Environment Linkages.¹

Agricultural systems (agroecosystems) are complex and intensively managed economic systems which produce a wide variety of economic food and fibre products, and provide other social, economic and environmental goods and services. Given their importance to national welfare, the maintenance of productive and sustainable agricultural systems is very much in the national interest, and an environmentally-sustainable agricultural industry is an official policy goal of the federal government (Agriculture and Agri-Food Canada, 1997). The agricultural economy and the broader environment are closely linked (see Figure 3). One way of describing this relationship is through the accounting concepts of stocks and flows, of which a generalized overview is

¹ This section draws on Smith and McRae (2000).

Figure 3: The agricultural economy and the environment

Source: Smith and McRae, 2000.

presented here.

Agroecosystems contain various resource stocks, some of which are used in the production process. Among these are agricultural land, soil, water, biodiversity, and various chemical elements, such as carbon, nitrogen, phosphorous, potassium and various micro-nutrients. Some of these stocks can be sub-divided into different classes. Land, for example, can be sub-divided by landscape form (based on factors such as slope, soil type, etc), by land cover (based on factors such as crop type or other form of land cover), or by soil capability (based on suitability for designated uses). Similarly, biodiversity can be sub-classified at the genetic, species (wild and domesticated) and ecosystem (habitat) levels. The aggregate of such resource stocks constitutes agriculture's natural capital (renewable and non-renewable), and their proper management and conservation are important criteria against which the sustainability of agricultural systems can be assessed.

Flows of resources can occur within the agricultural economy, between the agricultural economy and the environment, and between the agricultural economy and other economic sectors. On the input side, the environment provides many of the resources and conditions required for agricultural production, such as solar radiation, precipitation, land, soil, biological resources, and nutrients. Human-supplied inputs (such as fossil fuels, fertilizers, machinery, pesticides and management) are used to optimize outputs of economic commodities. Other outputs from agroecosystems are energy, nutrients and other materials (e.g. sediment) exported to other environmental systems (such as the atmosphere, groundwater, adjacent lands and adjacent surface water bodies). It is through these flows and cycles that agriculture is connected to the broader

environment, and external cycles of climate, water, and nutrients also affect agriculture. Examples of flows between agriculture and other sectors include energy inputs from the energy generation sector, and the energy embodied in commodities processed by the food and beverage industries.

Sustainability criteria related to flows concern the efficiency and impacts of resource consumption and the environmental impacts of output flows. For example, both overuse (i.e. inefficient use) of water by agriculture, and agricultural pollutant outputs (e.g. phosphorous runoff) into watercourses, may affect aquatic life and ecology. Factors which affect the environmental impacts of flows include the intensity of resource input use, the nature, magnitude, destination, and concentration of outputs in the receiving environment, and the sensitivity of the environment to damage. Furthermore, the significance of observed changes in stocks and flows must be assessed based on their degree of reversibility, scale of impact (e.g. area or population affected), relationship with a defined threshold (such as a water quality standard, a tolerable erosion rate or an accepted policy objective), and the benefits derived from the activity giving rise to observed changes, such as food production or economic gain.

Accounting convention holds that net changes in stocks should be accounted by patterns of flow. In principle, an accounting model of environmental resource stocks and flows within the agricultural economy, and between the agricultural economy, other economic sectors, and the environment, is possible. In practice, however, the development of such a model would be difficult, even conceptually, due to an imperfect understanding of the nature of agroecosystems and agroecosystem processes, and of

economy-environment interactions in agriculture. Indeed, the results of the review presented in Part B of this report suggest a comprehensive model of this type has yet to be developed specifically for agriculture.

10. Welfare-Enhancing Functions of Agricultural Land.

One way of organizing an environmental account for agricultural land is according to how such lands contribute to or affect welfare. In recent years, recognition and interest in the “multifunctional” character of agricultural land has grown (Food and Agriculture Organization, 1999), and because the functions performed or services provided can be valued in ecological and/or economic terms, they can be readily incorporated into an accounting approach, at least conceptually.

Policy issues may arise in cases where there is sufficient concern that the capability of land to perform socially-desired functions is jeopardized, or where tradeoffs between different functions exist and are perceived to be unacceptable. An agricultural land account should provide insight into whether the capacity of land to perform expected functions is being maintained, and where tradeoffs exist, a sense of the benefits and costs associated with the provision of various levels of functions or services. An overview of several key functions of agricultural land is presented in Table 12, and discussed in detail in the sections which follow.

Table 12: Welfare-Enhancing Functions of Agricultural Land	
Functions	Associated Issues
Crop production	<ul style="list-style-type: none"> - Retention of prime farmland - Enhancement of soil quality and productivity
Environmental services - Wildlife habitat - Carbon storage	<ul style="list-style-type: none"> - Management of adverse impacts of wildlife on farm production - Retention / enhancement of critical agricultural habitat availability and quality - Enhancement of net stocks of soil carbon to curb greenhouse gas emissions
Other environmental services	- None included in this study
Scenic landscape	- Retention of a scenic agricultural landscape
All functions	- Manage tradeoffs (e.g. productivity versus habitat retention or scenic value of land)

10.1 Crop Production.

Agricultural land has historically been, and continues to be, managed primarily for crop production. The contribution of such production to economic welfare is the main value captured through conventional economic accounting methods. The main factors which influence the capacity of land to perform this function are its availability and quality. Availability is a simple function of the amount and type of farmland retained for agriculture. Quality is the soil's fitness to support crop growth without resulting in soil degradation or otherwise harming the environment (Acton and Gregorich, 1995). Many factors influence soil quality, including inherent factors such as landscape form (e.g., slope), various soil chemical, physical and biological properties, and dynamic factors such as cropping patterns and land management practices (e.g. conservation practices).

Policy issues concerning this function relate to the retention (or loss) of highly-productive lands and their sustainable (or unsustainable) management and use.

10.2 Ecological Services.

The ability of agricultural land to perform several functions which can be labeled ecological services is increasingly recognized, and can be considered to be positive externalities (OECD, 1997b). Examples are the regulation and partitioning of water, the retention of soil in erosion-prone areas, the role of agricultural land in biogeochemical and other natural cycles, and the role of land as an environmental buffer (Acton and Gregorich, 1995). Accounting for such services facilitates analysis of how the capability of agricultural land to provide them compares with that of other land uses, such as land under “natural” cover or some type of “artificial” cover.

The economic and ecological importance of the ecological services provided by agricultural land can vary regionally. In Japan, for example, terraced rice paddy fields play a role in water storage, flood prevention, soil retention and erosion control in mountainous areas, and these functions have direct economic value. On the other hand, farmland in flat and sparsely populated areas, such as the Canadian prairies, would be a relatively less important provider of such services as flood control, but are a more important storage reservoir for carbon.

Historically in Canada, few significant policy issues have arisen concerning the supply (or under-supply) of ecological services from agricultural land. More recently however, interest in two specific ecological services of farmland has grown considerably: the provision of wildlife habitat and the sequestering of carbon in soil. The construction

of land accounts for these ecological functions may therefore be desirable, and is discussed below. Accounting for other ecological functions of land is likely to be of less value at this time and is not pursued further here.

10.2.1 Provision of Wildlife Habitat. Wildlife habitat includes all the things that a species needs to survive -- food, water, cover and home range (space). Habitat must also provide for special needs such as reproduction and dispersal, and different species may use different portions of the landscape to meet their resource needs.

In general, agriculture has reduced the quantity of natural habitat for wild species, mainly through the conversion of the natural landscape and changes in land use, such as draining of wetlands and removal and fragmentation of forest cover. However, agroecosystems can be a mosaic of cropland, pasture, woodland and wetland. These agricultural habitats, and this patchiness, can greatly benefit some wild species, and some are able to thrive where native habitat has been replaced by agricultural habitats. Relative to more developed areas, such as urban lands, agricultural lands can offer more habitat benefits to wildlife. A recent national assessment of the relationship between agricultural land and wildlife documented thousands of ways in which individual species of birds, mammals, reptiles and amphibians use agricultural land habitats in Canada (Neave et al., 2000). Agriculture can also act as a reservoir of domesticated plants and animals at the genetic and species levels.

Since agriculture occupies a substantial portion of the populous southern Canadian landscape, the wildlife habitat function of agriculture is increasingly valued by ecologists, conservation societies and the public. Key policy issues regarding this

function concern the adverse impacts of wildlife on farm production (e.g. grain consumption by migratory birds), and the retention of critical agricultural habitats in areas where wildlife is rare or threatened, or where wildlife using agricultural land provide economic benefits to landowners or other groups.

10.2.2 Carbon storage in soils. The role of agriculture in the carbon cycle has long been recognized: agroecosystems capture carbon dioxide from the atmosphere, bind carbon in organic form and store some of it in the soil. The agronomic benefits of enhancing soil carbon and soil organic matter are also well known. More recently, interest in the carbon storage or “sink” function of agricultural land has grown as the international community seeks ways of curbing the accumulation of greenhouse gases in the atmosphere. Several countries, including Canada, advocate including the sink function of agricultural land in the Kyoto Protocol as a way of offsetting emissions. The policy issue surrounding this function concerns how best to enhance net stocks of soil carbon in agricultural lands (Smith et al., 2000).

10.3 Provision of scenic landscapes.

The provision of a scenic and pleasing landscape is a potential function of agricultural land. For many people in densely populated countries (some European countries, for example), agricultural land provides the only semi-natural landscape readily available for aesthetic or recreational enjoyment. Similarly, the cultural features of such landscapes (such as alpine meadows, hedgerows, stone fences and historic buildings), which have evolved over centuries, help support a lucrative rural tourism industry. In some other countries however, such as Canada, scenic natural landscapes, such as

mountainous and forested areas, are more readily available and highly valued, thus the role of agricultural land in supplying scenic landscapes is (relatively) less important (OECD, 1998a). Nonetheless, many rural and urban Canadians value the visual beauty of the agricultural landscape, and in some parts of the country, such as Prince Edward Island, the rural and scenic character of this landscape underpins a strong tourism industry.

Clearly, this function remains a very subjective one that will vary considerably according to social values and preferences, and the relative availability of other desired landscapes. The main policy issue concerns finding a “proper” balance between the retention of scenic farmland (which tends to be small-farm oriented, mixed-crop and patchy) and enhancing agricultural productivity (which tends to favour large, more monoculture-based farming systems). This subjectivity impedes including this function in a land account for Canada pending resolution of definitional issues, and is not pursued further here.

11.0 A Functions-Based Approach to Agricultural Land Accounting.

An important function of an environmental account is as a system which contains the underlying data needed for various analyses, and the construction of relevant indicators. If a land account for agriculture is to serve this purpose, it should contain at least the following:

- contextual information on how agricultural land is being used and managed;
- information on whether the use and management of such land is undermining or enhancing its capacity to contribute to economic and ecological welfare (or stated

- otherwise, what benefits and costs result from the use of such land); and
- information on the exchange of materials and energy between land and other systems and/or sectors.

In this chapter, a framework agricultural land account is developed for this purpose. This account contains the following six sub-accounts: land use; land management; accounts related to the capacity of land for crop production, wildlife habitat, and carbon storage; and an account for exchanges of materials and energy between land and other systems and/or sectors. Each of these follows the basic accounting approach of recording levels and changes in stocks and flows using balance sheet and flow matrices, of which a generic illustration is provided for each in Tables 13 and 14.

Table 13: Generic Land Account Balance Sheet				
Region	Units	Opening Stock, Period, $t-1$	Closing Stock, Period t	% Change Over Accounting Period

The generic balance sheet presented in Table 13 can be applied flexibly. In principle, any study region (national, sub-national, etc) and accounting period (day, month, year, etc) can be specified according to the needs of the analysis or the properties of the object being studied, and stocks can be measured in either physical or monetary units. How land resource stocks are defined is also flexible. In the case of agriculture, for example, land stocks can be defined as total farmland, farmland by land use class (e.g. cropland, pasture) or farmland by quality class, depending on the function of interest.

Table 14: Generic Land Account Flow Matrix Period: t Region k					
From/To:	To/From: Units	Sector a or Stock i	Sector b or Stock j	Other stock or sector	Total gain/loss
Sector a or Stock i		X			
Sector b or Stock j			X		
Other stock or Other sector				X	
Total gain/loss					

Like the generic balance sheet, application of the generic flow matrix (Table 14) is also flexible. Both the accounting period and study region can be specified at any relevant temporal and spatial scale. The matrix tracks flows of stocks between stock classes, to other sectors, or from other sectors into stock classes. Total gains and/or losses by stock class or sector can readily be computed, and measurement is possible in either physical or monetary units. Again, the specification of appropriate stock classes and sectors will vary according to the specific land function of interest. The land accounting frameworks developed in the following sections are based on modified versions of these generic frameworks. Given the overall purpose of this research, macro-scale or regional scale information would be sufficient.

11.1 Accounting for Crop Production Capacity

As mentioned earlier, a land account for crop production should consider two factors: the availability (retention) of prime farmland; and the maintenance or enhancement of soil quality. An accounting framework is presented below for each factor.

Accounting classifications and definitions identified or proposed are based on those used in recent national studies applicable to Canadian agriculture.

11.1.1 Availability of prime (i.e. high inherent capability) farmland: A balance sheet for land stocks by crop production capability class is presented in Table 15. The balance sheet can be constructed for relevant time periods or spatial scales, and serves to indicate net gains or losses of high-capability farmland over the accounting period specified. In addition to recording overall changes in the availability of prime farmland, information on the flows of each land class between agriculture and other economic sectors is of interest. This can be achieved through a flow matrix table as presented in Table 16.

The completion of the balance sheet and flow accounts in Tables 15 and 16 depends crucially on the availability of meaningful land classifications for prime farmland, and an ability to track changes over time. Regarding land classification, the main national scheme available for this purpose is the Canada Land Inventory (CLI) (Environment Canada, 1978). This system classifies agricultural land into seven soil capability classes for agriculture, determined on the basis of inherent properties of soil which affect fertility and productivity. The first three classes are those considered suitable for annual crop production.

Table 15: Agricultural Land Account: Crop Production (Stocks)							
Factor: Land stocks by crop production capability class (balance sheet)							
Region	Unit	Land Class <i>i</i>			Land Class <i>j</i>		Land in class <i>n</i> .
		Period <i>t</i>	Period <i>t-1</i>	% Change	Period <i>t</i>	% Change	

Table 16: Agricultural Land Account: Crop Production (flows)							
Factor: Flows of Prime Farmland Period: <i>t</i>							
From/To:	To/From: Unit	Forest	Built-up land	Recreation	Other non-agriculture	Total gain/loss	
Land class <i>i</i>							
Land class <i>j</i>							
Other land class <i>n</i>							
Total gain/loss							

CLI maps of agricultural land capability have been georeferenced at the enumeration area level, which allows land use data collected through the Census of agriculture to be co-related to those maps. This in turn allows changes in land stocks by capability class to be tracked over time (Statistics Canada, 1997a,b). However, precise national information about the flows of such lands into and out of agriculture (i.e. their sources and destinations) is limited in Canada, making Tables 15 and 16 difficult to complete fully.

11.1.2 Soil quality for crop production: A balance sheet for land stocks by soil quality class is presented in Table 17. The balance sheet is identical conceptually to the one developed for prime farmland availability (Table 15 above), can be constructed for any relevant time period or spatial scale for which data exist, and serves to indicate net gains or losses of soil quality over the accounting period.

As with land capability, accounting for changes in soil quality depends crucially on the specification of soil quality parameters or classes. Recent national work on generalized assessment of soil quality for crop production is based on the concept of degradation risk, and is the approach used here to address soil quality. This approach combines information on the susceptibility of a soil to degradation (determined from inherent soil and landscape properties) with information on how a soil is being used and managed (based on prevailing cropping patterns and land management practices).

Examples of this approach are described briefly below. MacDonald et al., (1995, 1998) assign a ranking (classification) to soils based on their susceptibility to change and degradation. Key management factors considered are the % of farmland under crops, % of

cropland under row crops, and % farmland under summerfallow (prairie region only).

Another approach to generalized soil quality assessment (also based on the degradation risk method) is that used by Shelton et al., (2000), Padbury and Stushnoff (2000), King et al. (2000), McBride et al. (2000) and Eilers et al. (2000) for water erosion, wind erosion, tillage erosion, soil compaction, and soil salinization, respectively. Each of these studies assigns farmland to a specific degradation risk class for a given year, allowing for analysis of changes over time in the area or share of land in each class.

The assignment of land to a given class is based on an analysis of factors which affect specific degradation processes (e.g., soil & landscape properties, climate, and crop and management practices). To illustrate, the degradation risk classes for water erosion are:

1. tolerable (< 6 tonnes/ha/yr)
2. low (6 - 11 tonnes/ha/yr)
3. moderate (11-22 tonnes/ha/yr)
4. high (22 - 33 tonnes/ha/yr)
5. severe (> 33 tonnes/ha/yr).

All classes above the tolerable class are considered physically unsustainable.

Levels and changes of stocks of land in each soil degradation risk class can be tracked through the balance sheet approach presented in Table 17. Balance sheet accounts can readily be constructed for each degradation process. The development of an overall soil degradation risk index would allow for more integrated accounting, but such an index remains to be developed.

Table 17: Agricultural Land Account: Crop Production

Factor: Land stocks by soil degradation risk class (balance sheet)

[illegible]

Table 18: Agricultural Land Account: Crop Production

Factor: Flows of Land by Degradation Risk Class

Period: t ; Region k

From/to:	To/from: Unit	Degradation class i	Degradation class j	Degradation class n	Other degradation class	Total gain/loss
Degradation class i		X				
Degradation class j			X			
Degradation class n				X		
Other degradation class					X	
Total gain/loss						

With regard to changes in soil quality, flows between degradation risk classes are of interest as policy aims to shift farmland out of the unsustainable degradation risk classes into the tolerable classes. Table 18 demonstrates how a flow account matrix might be constructed to track such flows over time. Risk classes selected would be the same as those used in the balance sheet.

11.2 Accounting for Agricultural Wildlife Habitat.

An agricultural land account for wildlife habitat should consider the stock of agricultural habitat types available to support various wildlife and how these are changing over time (flows). An accounting framework is developed below for each factor. Compared to the role of agricultural land in crop production, this is a relatively new area of research.

11.2.1 Availability of agricultural habitats. If the stock of agricultural habitat is to be included in a national agricultural land account, a way of quantifying this habitat must be found. To do this, habitat types must be identified, and ways in which species use different habitats must be determined. This task is especially challenging since both agricultural habitat types, and the uses different species make of these, vary considerably within and across agroecosystems. Cropland, for example, may be used by different species as habitat in different ways, depending on the crop type grown.

Neave et al., (2000) developed an agricultural habitat-species matrix which identifies, by ecozone, how various species (of birds, mammals, reptiles and amphibians) use different agricultural habitats. Uses considered include breeding, nesting, feeding, foraging, wintering, and other uses. At the most general level, the habitat types were

defined to correspond to the land use categories reported in the Census of Agriculture (cropland, summerfallow, tame or seeded pasture, natural land for pasture, all other agricultural land), allowing this national data source to be used for analyses of agricultural habitat. Each of these was further sub-divided into more specific habitat types. Cropland, for example, is subdivided into area of each major crop (corn, small grains, etc), whereas All Other Agricultural Land is subdivided into farm wetlands, woodlots, and shelterbelts/fencerows. Each use by a species of a specific habitat type was recorded as a *habitat use unit* (i.e. the habitat use unit is not the number of species using a habitat, but the number of individual ways in which the habitat is used). To illustrate, Mallard feeding and Mallard nesting equal two habitat use units. This research confirms that all agricultural land has some habitat value, but the All Other Agricultural Land and Natural Land for Pasture census categories support the most habitat use units, followed by Cropland, Tame or Seeded Pasture, and Summerfallow (which is used little by wildlife).

Agricultural habitat might be quantified in an account balance sheet in two ways: by accounting for stocks of specific habitat types, and by relating changes in habitat types to the share of habitat use units which each habitat type supports. Table 19 illustrates a balance sheet account which tracks agricultural habitat using both methods. The account is completed here for illustrative purposes using data and classifications developed by Neave et al., (2000) for the agricultural prairie region.

A flow matrix showing changes in specific habitat types is presented in Table 20. Using this matrix, flows between habitat types can be tracked, such as would occur through the conversion of pasture to cropland. The flow matrix also allows for tracking of

exchanges between agricultural habitat and non-agricultural uses of land, information highly relevant to habitat conservation planners. For illustrative purposes, the broad habitat definitions and the habitat use unit concept used in Table 19 are also used in this Table, but no numbers are included as precise national data on flows between habitat types are sparse. As with the balance sheet in Table 19, the flow matrix allows for a summation of the % of total habitat use units whose habitat has either increased, remained constant or decreased during the accounting period.

Table 19: Agricultural Land Account: Wildlife Habitat Factor: Farmland habitat stocks (balance sheet) Region: <i>Prairie ecozone</i> Total habitat use units (HUUs): 3865							
Habitat Type	% HUUs	Habitat Stock (Mha)			Share (%) between 1981-96 of HUUs Supported By		
		1981	1996	% Change	Increased Habitat	Constant Habitat	Decreased Habitat
Cropland	17	18.76	22.2	18.33	17	---	---
Summer-fallow	.15	8.2	5.4	- 34	---	---	.15
Tame Pasture	4	1.98	3.6	81.8	4	---	---
Natural Pasture	19	10.8	10.0	- 7.4	---	---	19
Other Land	59	.81	2.1	160	59	---	---
Total	100	40.61	41.8	3	80	--	20

While the approach presented here identifies habitat availability within agriculture, a broader assessment would also consider other sources of habitat within the region of interest. If, for example, agricultural lands were the source of 80% of all

Table 20: Agricultural Land Account: Wildlife Habitat Function														
Factor: Farmland habitat flows Region: i ; Period: j ; Total habitat use units: k														
From/to:	% HUUs	To/From:					Habitat Type			Non- agri- culture	Net change	% of HUUs for Which Habitat has		
Habitat type		Crop- land	Summer- fallow	Tame Pasture	Natural Pasture	Other Land	In- creased	Re- mained constant	De- creased					
Crop- land		X												
Summer- fallow			X											
Tame Pasture				X										
Natural Pasture					X									
Other Land						X								
Non agri- culture									X					
Net change														

habitats, then these lands would be relatively more important for this function than were they to provide only 10% of available habitats. Contextual information of this sort could be recorded by adding a column to Tables 19 and 20.

11.3 Accounting for Agricultural Land Carbon. The dynamics of carbon fluxes in agricultural soils are complex and influenced by many factors, such as initial stocks of carbon, cropping and tillage practices, climate, and fertilization. Smith et al., (2000) report recent trends in soil organic carbon in agricultural soils at the national and sub-national level. They obtained estimates of soil stocks from Agriculture and Agri-Food Canada's soils database, and estimated rates of carbon change using the CENTURY model (a computer model designed for this purpose).

Table 21: Agricultural Land Account: Carbon Storage			
Period: t			
Carbon Stock	Units	Region i	Region n
Opening stock - total			
Additions - crop photosynthesis - manure - other - total additions			
Removals - crop exports - microbial respiration - erosion - other - total removals			
Closing Stock - total - rate of change			

The carbon storage function of agricultural land is, in principle, readily treated using the accounting balance sheet approach introduced in Table 13. Table 21 introduces a structure for doing this by accounting for opening stocks, additions, removals and closing stocks over the accounting period. Because additions and removals are shown in this table, no separate flow matrix table is proposed for this land function.

11.4 Accounting for Agricultural Land Management.

Agricultural systems are, by their very nature, managed systems. Management influences both the economic and environmental sustainability of agriculture. Poor management can lead to inefficient production, adversely affecting farm profitability. Such management can also result in environmental costs, such as degraded soils and excessive losses of materials and energy to the surrounding environment. Land management, then, has a direct bearing on the capacity of land to contribute to welfare as described earlier in this chapter.

A land management account can be used to track the level of adoption of those practices that contribute to the welfare-enhancing functions of land. In Canada and elsewhere, much research and policy effort is focused on developing and validating such practices for agriculture, and promoting their use. For example, the province of Ontario and Agriculture Canada published the *Best Management Practices* guidebook series for agriculture (Ontario Ministry of Agriculture, Food and Rural Affairs, 1992), and agriculture groups such as the provincial Soil and Crop Improvement Associations encourage their use on farms.

One important factor affecting the development of a land management account is

that recommended practices vary considerably, both regionally and across farming systems. The generic management account presented in Table 22 allows this variability to be considered. Its completion requires that the beneficial management practices be specified for the study area and/or farming system of interest.

Table 22: Agricultural Land Account: Land Management				
Region: i (balance sheet)				
Land Management Practice	Unit	A) Land Area treated by land type (e.g. cropland, total farmland) <i>and/or</i> B) Number of farms using practice by farm type (e.g., potato farms)		% Change Over Accounting Period
		Period _t	Period _{t+1}	
No-till Buffer strip Level of residue cover other	Ha or No. or other			

11.5 Accounting for Agricultural Land Use.

Land use is an important factor which can affect aspects of land such as its quality and productivity. Land mis-use occurs in cases where the uses made of lands exceed their inherent carrying capacity, thus leading to degradation. An agricultural land use account should include contextual information on how these lands are being used. Such information can serve as input into more detailed studies on agricultural land.

Agricultural land use can be classified in various ways and at different spatial scales, depending on study needs and available data. At the national level, the Census of

Agriculture provides the most readily available and consistent data set, with information and classifications based on the following aspects of land use:

- land area under various crops
- land area under summerfallow
- land area under pasture (improved, un-improved)
- agricultural land under other uses (an aggregate of land under woodlots, wetlands, laneways, buildings).

These land use categories provide a starting point for classifications for a national land use account for agriculture. Information could be supplemented through other sources, such as remote sensing. Tables 23 and 24 illustrate how an account balance sheet and flow matrix might be developed within an agricultural land account.

Table 23: Agricultural Land Account: Land Use (stocks)				
Region: <i>i</i>				
Land Use	Units	Period		
		<i>t</i>	<i>t + 1</i>	% Change
Cropland (total or by crop type)				
Summer-fallow				
Tame or Seeded Pasture				
Natural Land for Pasture				
Other Land				
Total Farmland				

Table 24: Agricultural Land Account: Land Use (flows)								
Region: <i>i</i> Period: <i>t</i>								
From/to	Units	To/From:					Non-agriculture	Net change
		Cropland	Summerfallow	Tame Pasture	Natural Pasture	Other Land		
Cropland (total or by crop type)		X						
Summerfallow			X					
Tame Pasture				X				
Natural Pasture					X			
Other Land						X		
Non-agriculture							X	
Net change								

11.6 Accounting for Agricultural Soil Stocks and Flows to Other Systems.

As mentioned earlier, agricultural land can also affect welfare through the exchange of materials with other sectors and resources. Examples of other resources or sectors include non-agricultural lands, aquatic systems and the atmosphere. The account framework presented in Table 24 would track such exchanges of land; thus what is of primary interest here are flows of other materials. The most obvious of these is soil, which may be lost from agricultural land through erosion or gained through natural soil formation or from erosion on adjacent lands not in agriculture. However, as described in chapter 9, there are exchanges of many other types of materials and energy between agriculture and other environmental and economic sectors (for example, of nutrients and greenhouse gases). A key question here then is whether an agricultural land account should include other such exchanges, or whether these are more properly addressed in an accounting framework that considers exchanges between the broader agricultural economy and the environment.

A more restrictive approach is taken here, focusing on transfers of soil between agricultural lands and other systems. In this context, soil loss or gain can affect agricultural productivity and income, and soil losses from agriculture to the environment can impose ecological and economic costs on society (as described and measured in several studies reviewed in Section 6.2) through sedimentation, eutrophication and other impacts, such as from pesticide residues. Tables 25 and 26 present, respectively, accounting frameworks for tracking stocks and flows of soil resources in agroecosystems.

Table 25: Agricultural Land Account: Soil Stocks			
Period: t			
Soil Stock	Units	Region i	Region n
Opening stock - total			
Additions - soil formation - transfers from adjacent lands - other additions - total additions			
Removals - erosion off-site - other removals - total removals			
Closing Stock - total			

Table 26: Agricultural Land Account: Soil Flows							
Period: <i>t</i>							
From/to:	To/from: Units	Agricultural land	Non-agricultural land		Aquatic Systems	Other systems	Total gain / loss
			sector <i>i</i>	sector <i>n</i>			
Agricultural land		X					
Non-agriculture land	sector <i>i</i>		X				
	sector <i>n</i>			X			
Aquatic systems					X		
Other systems						X	
Total gain/loss							

Table 25 requires that the term soil “stock” be defined. This could be an aggregate of all or selected soil types contained within a specified soil profile. A soil profile could be specified in several ways, depending on the needs of the accounting application. Options include by soil depth (e.g., 5, 10, 15 cm, etc) or soil horizon (e.g. A horizon, etc). Regarding soil flows in Table 26, it would be important to distinguish between erosion processes that redistribute soil on the landscape versus processes that move soil between landscapes. In practice, Table 26 would be difficult to complete nationally as little information is available on soil exchanges between agricultural lands and other lands.

12. Conclusion.

A conceptual accounting framework for agricultural land has been developed based on stock and flow accounting principles. In relation to the approaches to environmental accounting identified in Table 3, the agricultural land account proposed most closely resembles the “natural resource and environmental accounting approach”. Land stocks and flows are identified based essentially on physical properties of land and soil, and knowledge of how these interact with other systems. The integration of these accounts into a satellite account or the SNA itself is not reviewed here. Such integration would require economic valuation of land stocks and flows. Although selected methods for accomplishing this are reviewed in general terms in chapters 5 and 6 of this report, these are not discussed explicitly. As such, the land account proposed represents a first step towards a more developed economic/environmental account for agricultural land in Canada.

Despite this limitation, the account introduced here does offer a new and broader

perspective of Canadian agricultural land. By also recognizing and accounting for additional functions of such lands beyond their traditional economic crop production function, a more comprehensive picture emerges of how agricultural lands may contribute to economic and ecological welfare. Internalizing such functions in an accounting framework would increase the likelihood that such lands are managed for multiple functions, including crop production. The inclusion of soil stock and flow accounts (Tables 25 & 26) also sheds new light on how agricultural land use and management can affect other ecological systems and processes.

No attempt is made here to complete these accounts, but data available from existing national databases (e.g. Census of Agriculture) and studies (e.g.; McRae et al., 2000) could readily be adapted for such purposes, in particular for stock balances in physical terms. Completing the flow accounts, and accounting in economic terms, would likely be more difficult at the national level with existing data. As such, the accounting framework presented here may help identify key data gaps to be filled before a reasonably complete set of national agricultural land accounts can be developed.

Finally, the land accounts presented here should be seen as a subset of a larger set of accounts required to capture other linkages between the agricultural economy and the environment. In particular, a broader materials flow account would be required to track the consumption of resources other than land, and the generation and transfer of various pollutants, other than soil, by agriculture into the environment.

Part D: Case Study: Accounting for Stocks and Flows of Soil in the Grand River Basin of Ontario.

In this section, an aspect of the environmental accounting framework for agricultural land developed in Part C is applied to the Grand River watershed of Ontario (Figure 4). This relatively large watershed (some 680 000 hectares) flows into Lake Erie and consists of all the land that drains into the Grand River through tributary creeks and rivers. The Grand River basin was selected for this study because it is predominantly agricultural and because most (but not all) of the required agronomic, environmental and economic data are available.

This application develops a sub-account of the land accounting framework presented in Part C, specifically Tables 25 and 26 of section 11.6 which concern stocks and flows of agricultural soil. The analysis aims to account for levels and changes in cropland soil stocks, their interaction with adjacent aquatic systems, and resulting impacts on water users. Stocks of agricultural soils under cropland in the Grand River basin, and flows (transfers) of soil from this cropland to adjacent aquatic systems, are estimated in physical terms. Soil flows off cropland into water are also valued in economic terms, with valuation based on the damage costs of the resulting sedimentation of waterways.

General linkages between primary agriculture and water quality are first described in chapter 13, followed by a description of study methods and procedures in chapter 14. Chapter 15 presents the results of this study (detailed results are presented in Appendix A) and conclusions are summarized in chapter 16.

Location of Soil Landscape Polygons within the Grand River watershed

13. Agriculture and Water Quality.

Water quality is usually defined as a function of its intended use. Uses of water vary considerably from, for example, drinking (by humans or livestock), recreation, irrigation and as habitat for aquatic life. Methods used for measuring water quality have often taken a chemical approach, and the Canadian Environmental Quality Guidelines specify acceptable and unacceptable levels of contaminants in relation to such uses. In addition to certain chemicals, water quality guidelines also exist for biological and physical contaminants, such as bacteria, algae, sediment levels, turbidity, and temperature (Canadian Council of Ministers of the Environment, 1999). Water quality for use, k , can therefore be characterized as a function of an n -dimensional parameter vector as follows:

$$Q_k = f(p_{jk}) \quad (j = 1, \dots, n) \quad (1)$$

where Q_k is the water quality vector for use k , and p_{jk} is the observed value of the j th parameter in use k . In this relationship, if the observed value of any of the n parameters exceeds its specified threshold (i.e. water quality guideline) level, the water is considered unsuitable for use k (van Vuuren et al., 1997).

The relationship between agriculture and water, including water quality, is comprehensively described in Coote and Gregorich (2000). Agriculture is the largest consumptive user of water of all economic sectors in Canada, and can affect the quality of this resource through the discharge of several contaminants. The main agents of declining water quality that stem from primary agriculture are soil particles, crop nutrients, pesticides and bacteria. Each is briefly described below, drawing on material presented in Harker et al., (2000).

“Sediments. Sediments are mainly composed of soil particles that enter water from erosion. Much of the eroded soil that enters water remains in suspension, affecting the turbidity of water and its suitability as habitat for aquatic life. Sedimentation may also reduce the storage capacity of reservoirs and wetlands, clog drainage ditches and irrigation canals, affect navigation channels and many other economic uses of water. Sediments from farmland may also transport other agricultural contaminants into water, such as nutrients, pesticides, and pathogens, thus affecting water fitness for various uses.

Nutrients. Nutrients are chemical elements, such as nitrogen, phosphorus and potassium, that are needed by plants for growth. These occur naturally but are also added to agricultural land to optimize yields, mainly in the form of mineral fertilizers, animal manures, plant residues and the plowdown of legumes. If the amount of such nutrients in the soil exceeds the amounts taken up by the crop, there is a risk that surplus nutrients will move off farmland into adjacent waters, where elevated levels can affect water quality, such as through eutrophication.

Pesticides. Pesticides are applied to farmland to control weeds, insects, and diseases that interfere with agricultural production. Many pesticides are applied to soil, where they can be transported off farmland into surface waters through runoff, or leach through the soil profile into tile drainage water or into groundwater. Safe limits for pesticides are not always known, and their presence can affect water quality for aquatic life, for drinking, and for other uses.

Pathogens and other contaminants. Pathogenic organisms, including bacteria, viruses and parasites, occur naturally in soil and water, but high levels can affect uses of water, in particular for drinking and recreation. Other agricultural contaminants that can impair water quality are heavy metals, whose entry into water can pose a risk to humans and aquatic life, and endocrine-disrupting chemicals, which may affect humans, fish and wildlife using water that might be contaminated” (Harker et al., 2000).

14. Study approach.

The overall approach used in this study is illustrated in Figure 5. Stocks of soil, and soil losses from agricultural lands to aquatic systems, are estimated in physical terms. Economic uses made of water, and the value of damages imposed on water users by a unit of pollution (soil sediment), are then determined. Finally, the economic value of damage costs due to soil flows from cropland into water are estimated at the basin and Soil Landscape Polygon levels. Specific study procedures are described below.

Figure 5: Environmental accounting approach used in this study

14.1 Estimating soil stocks (Figure 5, Step 1). Accounting for different stocks, flows and impacts requires that these be expressed on a common basis for comparative purposes, and the unit “tonnes of soil per hectare” is used as such here. To estimate the total stock of soil in the study area, a factor to express soil weight on an areal basis is therefore needed. There are several ways of measuring soil weight, such as on a particle density basis (the mass of a unit volume of soil solids), or on a bulk density basis (defined as the mass of a unit volume of dry soil). Whereas particle density is concerned with the soil particles only, bulk density is concerned with the quantity of pore spaces as well as soil solids (Brady, 1974).

Brady (1974) provides several weight factors for soils. On a particle density basis, most mineral soils usually vary between 2.6 and 2.75 g/cm³. For bulk density, clay, clay

loam and silt loam surface soils normally vary from 1.0 to 1.6 g/cm³, and sands and sandy loams vary from 1.2 to 1.8 g/cm³. Brady also provides an average factor for the actual weight of a soil as being between 2 - 2.5 million pounds per acre to a depth of 6-7 inches. Comparable weight figures expressed in metric units are 2 200 to 2 800 tonnes per hectare in the upper 15 cm of the soil profile. In this study, the average of this range is used - 2 500 t/ha to 15 cm of soil depth. A depth of 15 cm was selected as this represents the agronomically active soil layer, and the stock of cropland soil to this depth was calculated using equation 2².

$$S_t = (\text{area of row crops}_t) (2500 \text{ t/ha}), \text{ where} \quad (2)$$

S_t = the stock of cropland soil in tonnes to 15 cm depth and time t

$\text{area of row crops}_t$ = hectares under row crops at time t

Changes in soil stocks are a net of additions minus removals over the accounting period. In this application, only additions of soil through soil formation are included; data on imports of soil into crop lands from other lands are lacking, but likely of minor importance. The natural rate of soil formation rate in Ontario has been estimated at 0.65 t/ha/yr, based on evidence that about 60 cm of soil has formed over the past 12 000 years (Kettlewell and Groenvelt, 1996). However, Kettlewell and Groenvelt (1996) distinguish “natural” soil formation from “accelerated” soil formation, the latter occurring when soils are actively worked, vegetated and mixed, leading to higher rates of soil formation.

² It is important to note that this calculation does not estimate the total stock of soil, which can be defined as all soil present in the soil profile above the parent material – typically the O, A and B horizons – which are collectively referred to as the Solum. The total stock of cropland soil within and across SLCs would thus vary by the depth of the prevailing soil profile.

Accelerated soil formation is thus more representative of conditions where soil is worked regularly, so the rate of natural soil formation likely underestimates the actual rate of soil formation on cropland in southern Ontario. The selection of an appropriate soil formation factor for southern Ontario cropland was discussed with Dr. Greg Wall, who suggested a doubling of the natural rate of soil formation factor would be more representative of present conditions (G. Wall, personal comm., 2000). A factor of 1.3 t/ha/yr is therefore used in this study to estimate soil formation on cropland. Losses of soil to water were estimated using the procedure described in section 14.2 below.

14.2 Estimating flows of soil from cropland into water (Figure 5, Step 2). Soil losses (sediment yields) into watercourses represent a loss of soil from agriculture, and a net loss if the actual loss exceeds additions through imports from other systems and through natural soil formation. As mentioned above, any transfer of soil particles from land into water represents a potential ecological and/or economic cost due to reduced water quality.

There are several ways through which flows of soil from cropland into watercourses might be estimated. The sediment yield of a basin is a function of several factors, including climate, basin geomorphology, soil type, and land use (Dickinson and Green, 1988). Much of the initial work on non-point pollution of the Great Lakes from agricultural (and other) sources was conducted through the Pollution from Land-Use Activities Reference Group (PLUARG) study (PLUARG, 1978). For example, Wall et al., (1982) calculated sediment yields from agricultural cropland in several Southern Ontario watersheds (including the Grand) using three different, independent methods. The first method extrapolates actual soil particle data in the water column obtained from

smaller agricultural watersheds to other areas of similar soil, landscape and climatic regimes. The second method uses the Universal Soil Loss Equation (USLE) to calculate potential soil erosion rates (based on prevailing landscape, soil, climatic and cropping conditions), to which a gross delivery ratio factor was applied (obtained from the USDA Soil Conservation Service) to predict sediment yields. The third method utilized a regression equation developed by Wall et al., (1982) using generalized watershed information and actual measured sediment yields to predict suspended sediment loads (the watershed variables included % clay and % sand of soils, available P, fertilizer P, manure, total P added, percent hay and pasture, alfalfa, woodland, row-crops, animal units, rural residents and stream density). The developed equation is as follows:

$$Y = -204 + 7.9 (\% \text{ row crop}) + 11.0 (\% \text{ clay}) \quad (3)$$

where Y is the predicted soil loss (kg/ha/yr) into water.

Other studies have used other methods to predict soil losses from cropland. Several of these use the GAMES model (Guelph model for evaluating the effects of Agricultural Management systems on Erosion and Sedimentation). The approach used in this model involves the disaggregation of a watershed into field-sized elements with homogenous characteristics of land use, soil type and class of slope. Gross erosion rates for each element are calculated using the USLE. The amount of soil which is delivered from each cell to the watershed's stream channels is determined by a calculated delivery ratio for each cell. This ratio is based on physical characteristics of the cell, such as roughness, slope, and flow length based on proximity of field to stream (Rudra et al., 1986). The GAMES model requires detailed input data and has been used mainly at the

scale of small watersheds. For example, Fox and Dickson (1990) used GAMES to estimate soil losses into water in three sub-watersheds in the Thames River basin (Big Creek watershed - 3300 ha; Newbiggen Creek watershed - 2647 ha; Stratford/Avon watershed - 537 ha). Fox et al., (1995) used GAMES for the same purpose in two of these three watersheds (Stratford/Avon and Big Creek), and van Vuuren et al., (1997) used GAMES in two sub-watersheds within the Kettle Creek watershed.

The equation developed by Wall et al., (1982) (equation 3 above) is used in this study, for the following reasons. First, it was developed and validated using detailed, measured sediment yield data obtained from agricultural watersheds in Southern Ontario. Second, the equation is readily applied and calculated at the broad, generalized spatial scale used here, and the required data are available from Agriculture and Agri-Food Canada's national soils database. Third, the lead author of the study that developed and applied the equation, Dr. Greg Wall, felt the equation remains valid for application in southern Ontario (G. Wall, pers. comm., 2000). Independent discussions with Dr. Dick Coote confirmed this view (D. Coote, pers. comm., 2000). The other two methods for estimating soil losses into water were considered but rejected. The use of gross sediment delivery ratios was tried and ultimately rejected by Wall et al., (1982), who concluded this approach did not appear appropriate for Southern Ontario. Use of the GAMES model at so large a scale was deemed impractical for this study due to the data requirements of so large a study area.

There are, however, limitations associated with use of this equation. As explained in Wall et al., (1982), the equation explains 64% of the total variation in the measured

sediment yield data ($R^2 = 64$). While the equation implicitly takes landscape factors such as slope into account, more explicit accounting of such characteristics, as is done in the GAMES model, might improve the accuracy of the predictions. Finally, the equation does not take conservation practices that reduce erosion, such as reduced tillage and riparian buffer strips, into account. No data are available on the area of buffer strips used in the Grand River watershed. However, data on tillage are available from the census of agriculture, and these show that tillage practices in Ontario have evolved considerably over the past 20 years (Statistics Canada, 1997c), with erosion-reducing effects. Between 1981 and 1996 for example, about 7% of Ontario's cropland shifted into the tolerable risk class (gross erosion below 6 tonnes/ha/yr) from higher erosion risk classes, mainly due to reductions in tillage (Shelton et al., 2000). However, this calculation only considers erosion on the agricultural landscape, which may or may not represent reductions in soil losses into water.

It is possible that recent changes in cropland residue associated with different tillage regimes would not have that great an effect on overall soil losses to water. Most reduced tillage implementation has been on coarse to medium textured soils rather than fine (clay) soils where sediment production and transport are highest. Thus while the coefficients of equation 2 might change slightly the overall relationship would stay intact (G. Wall, pers. comm., 2000). It is also possible, however, that reduced tillage will lead to reductions in soil losses from cropland to water, and several studies point to such an effect. Using the GAMES model, for example, Fox and Dickson (1990) estimated that conservation tillage and no-till reduced the volume of soil delivered into water by about

41% and 77% respectively in three watersheds of southern Ontario. Comparable reductions, also using the GAMES model, were also estimated by van Vuuren et al. (1997) in southern Ontario. In the present study, soil losses from cropland into water are estimated with and without taking conservation tillage practices into account.

To run the sediment yield equation, three factors were obtained from Agriculture and Agri-Food Canada's national soils database (NSDB): 1) delineation of spatial units for the calculations, and, for each unit, 2) an estimate of the % clay content of soils and 3) percent of total agriculture area under row crops. The spatial units employed in the NSDB are Soil Landscape Polygons (SLCs) mapped at a scale of 1:1 000 000 (at this scale, the smallest parcel of land that can be displayed is about 2 500 ha). Data on various soil and landscape properties, and data from the national Census of Agriculture (collected at the enumeration-area level) are available by SLC, allowing for the integration and analysis of physical, economic and farm management data.

There are some 3123 SLCs in Canada with at least 5% or more of their area in agriculture (Smith and McRae, 2000). However, SLC boundaries do not correspond to watershed boundaries, so a special procedure was needed to match the Grand River watershed boundary with the SLC polygon boundaries. To achieve this, the ARC/INFO geographic information system's Union function was used to overlay the relevant portion of a watershed/sub-watershed boundary file developed for the Great Lakes region (file INF 004 igc obtained from Agriculture Canada) with a land cover file from version 1.9 of Agriculture Canada's SLC database. ARC/INFO's Frequency function was then used to identify the proportion of SLCs falling partially or wholly within the watershed. Thirty-

nine SLCs, representing some 1.75 million hectares (Mha) of land, were identified that lie either wholly or partially within the watershed (see Annex Table A1 and Figure 4). Only 8 of the 39 SLCs fall entirely within the watershed; for the remainder the proportion of each SLC's area falling within the watershed was estimated. Using this approach, 690 811 ha of land, and 476 164 ha of farmland, were identified within the Grand River watershed. The Grand River Conservation Authority identifies the total watershed as occupying some 680 000 ha of land, which represents 98.4% of the estimate obtained from the procedure used in this study.

The % row crop factor was calculated as follows. Row crops were defined to include vegetables; tobacco; potatoes; sugar beets; corn (grain and silage); soybeans and other beans, peas and lentils; and small grains (wheat, oats, barley and other annual row crops such as canola, millet, etc.). Row crops such as berries, grapes and tree fruits were excluded since these are managed to retain soil stability between rows (through such techniques as mulching). Area under row crops was then summed and divided into total farmland area for each SLC (see Annex Tables A2 and A3). It was assumed that the row crops are uniformly distributed within each SLC.

A representative value for the percent clay content of soil in each SLC was obtained from the version 2.1 component table and carbon layer table of Agriculture and Agri-Food Canada's national soils data base (see Table A3). Data were generated using the texture_clyr file to convert from texture class to percent clay. The data for the top 15 cm of all mineral soils were extracted (the same soil depth used to estimate soil stocks as described in section 14.1 above), and a weighted average (by depth of the layers) of the

clay content was determined for each component. The spatially weighted average of those clay contents was then determined for each polygon. Clay content is as a percentage of the mineral soil not including coarse fragments.

To account for tillage, data on the proportion of seeded area under each of conventional, conservation and no-till tillage were obtained from Agriculture and Agri-Food Canada's national soils database for each SLC for 1991 and 1996. This factor was then used to estimate the area of row crops under each tillage regime (see Table A4). This procedure assumes that the tillage regimes are uniformly distributed across the row crops, and that all tillage in 1981 was conventional tillage (this assumption is reasonable since conservation tillage technology was little developed in 1981). It was also necessary to equate the definitions of tillage types used by Fox and Dickson (1990) with those used on the Census of Agriculture (which is the source of the cropping and tillage data contained in AAFC's NSDB). Fox and Dickson defined conservation tillage as fall chisel ploughing. Discussions with Greg Wall (personal comm., 2000) and Catherine Cromey (personal comm., 2000) confirmed the validity of equating this to conservation tillage as defined in the Census of agriculture. They also confirmed that ridge planting as defined by Fox and Dickson (1990) equated with no-till.

Soil losses from agricultural cropland to water in the basin were estimated as follows. Losses for each SLC (Y) were calculated using equation 3. Losses in kg/ha/yr for each SLC were then converted into total annual tonnes of soil loss using equation 4. This number was then adjusted to account for the proportion of each SLC falling within the watershed using equation 5.

$$Y_{SLC\ ty-1} = (Y) (\text{ha row crop in each SLC}) (0.001), \text{ where} \quad (4)$$

$Y_{SLC\ ty-1}$ = tonnes of soil loss / yr from agricultural land under row crops in each SLC

Y = kg/ha/yr of soil loss in each SLC (obtained from equation 3)

$$Y_{GR\ SLC} = (Y_{SLC\ ty-1}) (\% \text{ of SLC lying within study area}), \text{ where} \quad (5)$$

$Y_{GR\ SLC}$ = tonnes/yr of agricultural soil loss into study area from each SLC.

The final step involved summing the estimated soil losses from each SLC in the study area to obtain a total soil loss estimate for the full watershed (equation 6).

$$Y_{GR} = \sum Y_{GR\ SLC}, \text{ where} \quad (6)$$

Y_{GR} = tonnes of agricultural soil particles deposited annually into waters of the Grand River watershed.

Results for 1991 and 1996 were also adjusted to account for changes in tillage from the 1981 base year. For these years, the “ha row crop” factor in equation 3 was subdivided into three categories representing hectares under conventional, conservation and no-till. Soil loss reduction factors obtained from Fox and Dickson (1990) were applied to the latter two tillage regimes, as follows:

- for hectares under conservation tillage: soil loss was multiplied by 0.593
- for hectares under no-till, soil loss was multiplied by 0.227.

These factors were derived from three case studies completed in watersheds in southwestern Ontario where agriculture is also a dominant land use. While it is important to note they were not derived from field observations in the Grand River watershed, there is sufficient similarity between these watersheds and that of the Grand River to suggest they are a transferable proxy that can be used in this analysis.

14.3 Estimating economic damages from soil losses into water. Soil particles deposited into watercourses represent an externality of agricultural production. The economic value of this externality depends on the degree to which such soil losses adversely affect consumer or producer utility. Any reductions in the level of damage over time would increase consumer or producer utility, and thus constitute a benefit. Agricultural producers derive utility from soils through their contribution to economic crop production. Consumers derive utility from water in multiple ways, including directly for drinking, recreation (bathing and fishing), and flood control, and indirectly through uses by industry, such as water storage for irrigation and power generation, for transportation, or for industrial cooling processes. Water also has existence or amenity value.

The external (off-farm) costs of agricultural soil losses into water are not fully captured by existing economic accounting procedures. This analysis accounts for such costs to water quality resulting from primary agriculture for 1981, 1991 and 1996. Accounting for them requires that changes in consumer utility due to soil losses into water be estimated, and several studies have tabulated such costs (or partial costs). For example, Ribaud (1989) compiled cost estimates for regions of the United States (see section 6.2.2 for a summary of how these estimates have been used in the US) by considering damages to the following: freshwater and marine recreation, water storage capacity, navigation, flooding, roadside ditches, irrigation ditches, commercial fishing, municipal water treatment, municipal and industrial water use, and steam power cooling. Other researchers have estimated or applied similar damage costs in parts of southern Ontario. van Vuuren et al., (1997), for example, used damage cost estimates for two water

uses in the Kettle Creek watershed: angling and water conveyance. They obtained average estimates of anglers' willingness-to-pay for water quality improvements from a Washington State study (Willis et al., 1992), and used estimates of water conveyance costs derived by Fox and Dickson (1990). Fox and Dickson (1990) also develop estimates of average annual costs to recreational angling and water treatment per tonne of sediment from agriculture.

The valuation approaches used here are consistent with the contingent valuation approach recommended by the United Nations (1993) and the social cost of pollution approach recommended by Hamilton and Ward (1998) for valuing environmental services for environmental accounting purposes (described in section 5.3). Ideally, the economic damage costs (or benefits of reduced damages) per tonne of soil particles deposited into waters of the Grand River basin would be used, but such estimates are unavailable. It is thus necessary to use estimates developed for other regions as proxy values. To minimize the risk of introducing error associated with using data from other locations, the values used should be derived from watersheds with socio-economic and environmental conditions similar to those of the study area. In practice, this means the stressors (i.e. factors causing damage), water uses, and types of damage considered should be similar in both the study area and the area(s) from which the proxy damage values are obtained.

The principal economic uses made of water in the Grand River basin were identified as follows (Figure 5, Step 3):

- recreational angling: personal or through commercial outfitters
- other recreation, such as canoeing, rafting and some swimming in reservoirs

- municipal water use for households and industry
- water storage, through some 32 dams built mainly for flood control with some power generation
- irrigation, mainly for golf courses with some for agricultural crops (Mr. Joe Farwell and Ms. Laurie Minshall, personal comm., 2000).

To identify damage costs due to soil losses that could serve as proxies for the Grand River basin (Figure 5, Step 4), the following studies were reviewed --

Ribaudo, 1989; Fox and Dickson, 1990; Fox et al., 1995; and van Vuuren et al., 1997.

This review revealed that the latter three studies mainly used similar cost data, thus this study draws on two data sets: those of Ribaudo and a consolidation of those of the other studies identified above. However, van Vuuren et al., (1997) differed by estimating the benefits to angling of reduced soil losses from a Washington State study by Willis et al., 1992, so the angling values used by van Vurren et al., (1997) are not used here.

Ribaudo (1989) developed damage estimates for various water uses for 10 regions of the US. Of these, estimates from the US Lake States and Northeast regions are shown here as these regions most closely approximate conditions in southern Ontario (Table 27).

Damage costs per tonne of soil particles are based on estimates of 65.74 and 90.47 million tonnes of soil particles discharged into watercourses from cropland for each region, respectively, as reported in Ribaudo (1989).

Table 27: Annual offsite damage from soil particles, million 1986 dollars ³						
Damage category	Lake States, US\$	\$/tonne		Northeast States, US\$	\$/tonne	
		US \$	Cdn \$		US \$	Cdn \$
Freshwater recreation	189.162	2.88	4.00	336.350	3.71	5.16
Water storage	52.666	0.8	1.11	18.729	0.20	0.28
Navigation	29.954	0.45	0.63	65.131	0.72	1.00
Flooding	65.110	0.99	1.38	103.119	1.14	1.58
Roadside ditches	22.584	0.34	0.47	50.017	0.55	0.77
Freshwater commercial fishing	15.424	0.23	0.32	0.220	0.001	.003
Municipal water treatment	113.695	1.73	2.40	183.213	2.02	2.81
Municipal & industrial use	176.053	2.67	3.70	193.679	2.14	2.97
Steam power cooling	4.737	0.07	0.1	33.05	0.36	0.51
Total damages	669.384	10.16	14.11	983.508	10.87	15.10

Source: Ribaud, 1989 and personal communication.

Table 28 summarizes soil particle damage costs reported for southern Ontario. These estimates, and the methods used to generate them, are summarized in Fox and Dickson (1990) so only a brief summary is provided here. For recreational angling, the total benefits resulting from the elimination of excess soil particles from streams and lakes in southern Ontario were estimated to be about \$35 million. This value was divided into Wall et al.'s, (1982) estimate of 652,872 tonnes per year of total soil delivered from

³ US currency was converted into equivalent 1986 Canadian currency using a factor of 1.3894 obtained from the Bank of Canada.

cropland and streambanks in the Ontario portion of the Great Lakes basin, giving an average loss to recreational angling of \$52.92 per tonne of soil particles per year. For water treatment, an annual average cost of \$13.44 per tonne of soil particles was derived using data from southern Ontario on gallons treated and operational costs of treatment per gallon. The proportion of this cost attributable to cropland erosion was estimated using factors obtained from the US EPA (1979) and Barbos et al., (1987) to be about 5% of total operational costs (\$10.2 million per year). This number was then divided by Wall et al's., (1982) estimate of total soil particles delivered in southern Ontario. For soil particle buildup in drainage ditches, Fox and Dickson estimated damage costs to be \$3.41 per hectare of cropland.

Table 28: Benefits from reduced soil losses into water in selected basins of southern Ontario				
Watershed	Beneficiary	Annual Value (1987 \$)		Units
		Average	Range	
Big Creek	Recreational angling	79.4	---	\$ / tonne / year
	Water conveyance	2.27	3.41 - 1.14	\$ / ha / year
	Municipal (Mun.) water treatment	20.16	---	\$ / tonne / year
Newbiggen Creek	Recreational angling	52.93		\$ / tonne / year
	Water conveyance	2.29	3.41 - 1.18	\$ / ha / year
	Mun. water treatment	13.44	---	\$ / tonne / year
Stratford/A von	Recreational angling	26.47	---	\$ / tonne / year
	Water conveyance	2.37	3.41 - 1.28	\$ / ha / year
	Mun. water treatment	6.72	---	\$ / tonne / year

Source: Fox and Dickson, 1990.

The damage estimates shown in Tables 27 and 28 differ considerably, with Ribaudo's per tonne damage estimates being much lower for comparable water uses. It is not possible to precisely account for these differences here: they possibly reflect differences in the economic uses and value of water, in the nature and cost of treatment technologies used, and/or differences in the valuation methods used. It is also possible that the Canadian studies, which focus on much smaller areas, were able to access more comprehensive and detailed data, leading to high damage cost estimates.

Table 29 presents the damage cost estimates used in this study in 1996 Canadian dollar values (Step 4, Figure 5). These estimates were derived as follows. The damage estimates reported in Fox and Dickson (1990) for recreational angling and municipal water treatment are used as these are derived from Ontario data and relate to important water uses in the Grand River basin. To these are added Ribaudo's damage estimates for water storage, flooding, and ditch maintenance, since these are costs likely to be incurred from sedimentation in the Grand River basin, and this source represents the best available alternative as no Ontario data are available. Ideally, damage estimates would be adjusted by SLC as required, but as this is not possible it is assumed that these damage costs apply uniformly throughout the basin.

For those estimates obtained from Ribaudo, the higher value reported for either the Lake States or Northeast States region was selected (since Ribaudo's estimates are considerably lower than comparable estimates from the Canadian studies). All figures were converted to 1996 dollar values using the Bank of Canada's Consumer Price Index Inflation Calculator.

Table 29: Annual damage estimates per tonne of soil particles used in this study		
Damage category	Value (1996\$)	Source
Municipal water treatment	17.60	Fox and Dickson, 1990
Recreational angling	69.32	
Water storage	1.51	Ribaud, 1989
Flooding	2.16	
Water conveyance (ditch maintenance)	1.05	
Total	91.64	

The final step (Figure 5, Step 5) involved estimating the economic costs of soil losses into water, and the economic benefits to water quality (measured as damages avoided) from the use of reduced tillage techniques. Equation 7 was used to calculate the annual damage costs from soil losses into waters of the Grand River basin per SLC. These costs were then summed for the entire basin using equation 8.

$$DC_{GR\ SLC} = (Y_{GR\ SLC}) \times (DC_{sT}), \text{ where} \quad (7)$$

$DC_{GR\ SLC}$ = annual damage costs from soil losses into waters of the Grand River basin per SLC

$Y_{GR\ SLC}$ = tonnes/yr of soil losses into waters in the Grand River basin from each SLC

DC_{sT} = annual dollar damages per tonne of soil loss into water

$$DC_{GR} = \sum DC_{GR\ SLC}, \text{ where} \quad (8)$$

DC_{GR} = total annual damages to water quality in the Grand River basin

$DC_{GR\ SLC}$ = annual damage costs from soil losses into waters in the Grand River basin per SLC

Equations 9 and 10 were used to estimate the economic benefits to water quality of reduced tillage by SLC and for the full basin.

$$B_{RT\ GR\ SLC} = (Y_{GR\ SLC\ CT} - Y_{GR\ SLC\ RT}) \times (DC_{\$T\ H}), \text{ where} \quad (9)$$

$B_{RT\ GR\ SLC}$ = economic benefits to water quality of reduced tillage per SLC

$Y_{GR\ SLC\ CT}$ = soil losses into water per SLC in the Grand River basin under 100% conventional tillage

$Y_{GR\ SLC\ RT}$ = soil losses into water per SLC in the Grand River basin under prevailing tillage practices

$DC_{\$T}$ = annual dollar damages per tonne of soil loss.

$$B_{RT\ GR} = \sum B_{RT\ GR\ SLC}, \text{ where} \quad (10)$$

$B_{RT\ GR}$ = total annual benefits to water quality in the basin from reduced tillage

$B_{RT\ GR\ SLC}$ = economic benefits to water quality of reduced tillage per SLC.

15. Study Results.

15.1. Stocks of soil under cropland in the study area. Estimates of soil stocks under cropland to 15 cm of soil depth are presented in Table 30 for three accounting periods: 1981, 1991 and 1996. These results suggest that soil losses *into water* from cropland pose no significant threat to cropland soil productivity in the watershed. These losses represent only a fraction of the total available stock which, overall, is near equilibrium ⁴ (the overall reduction in opening soil stocks to 15 cm of depth from 1981-1996 is due primarily to a slight reduction in total cropland area). However, gross erosion rates on cropland would

⁴ Although the *total* stock of soil is not estimated here, the fact that additions of soil slightly exceed soil losses to water suggests the total stock is near equilibrium.

be higher, and additional research would be needed to determine whether soil productivity is being affected by a re-distribution of soil within the agricultural landscape.

Table 30: Balance sheet of cropland soil stocks in the Grand river watershed under prevailing management practices, 1981, 1991, 1996 (tonnes)			
Stock	1981	1991	1996
Opening stock (to 15 cm depth)	651 538 250	604 266 250	640 244 525
Additions (from soil formation)	338 800	314 218	332 927
Removals (to watercourses) ⁵	130 535	106 494	103 470
Net change (total)	208 265	207 724	229 457
Change (as % of opening stock to 15 cm depth)	+ 0.0003	+ 0.0003	+ 0.0003

15.2 Soil losses from cropland into water. Summary results of the soil loss calculations are presented in Figures 6 and 7, and Tables 31 and 32. Detailed results are presented in Appendix Tables A5 - A9, and A13, by individual SLC and for the entire Grand River basin, in both total tonnes per SLC and tonnes/ha of cropland. Results are presented spatially in Figures 8 and 9, and in Appendix Maps A1 and A2. Soil losses are estimated for two tillage scenarios (100% conventional tillage, and prevailing tillage practices).

If all tillage is assumed to be conventional over the three accounting periods, mean soil losses per hectare of cropland for the basin decreased from 0.46 tonnes in 1981 to 0.45 tonnes in 1996, a reduction of approximately 3% (Figure 6; Table A5). Measured

⁵ See section 15.2.

in terms of total tonnes of soil loss for the entire basin, the reduction from 1981 to 1996 is even smaller, about 1% (Figure 7; Table A6).

When prevailing tillage practices are taken into account, soil loss estimates differ considerably. At the basin level, total soil losses dropped by 18%, 3% and 21% between 1981-1991, 1991-1996 and 1981-1996, respectively (Figure 7, Table A7). On a mean t/ha basis, soil losses were reduced by 15%, 11% and 25% for 1981-1991, 1991-1996, and 1981-1996 respectively (Figure 6, Table A8).

Tables A9 and Figures 6 and 7 show soil loss estimates for 1991 and 1996 with and without considering tillage effects. Adoption by farmers of conservation and no-till practices reduced total soil losses into water from cropland by 10% and 20% in 1991 and 1996, respectively, from what otherwise would have been the case under conventional tillage.

The spatial distribution of soil losses within the watershed in 1996 is summarized in Table 32 and shown in Maps A1-A2. Five SLCs, representing 56% of row crop area and 48% of farmland, accounted for 60% of total soil losses from cropland into the watershed. Generally, those SLCs with the largest total area within the watershed contributed the largest amounts of soil to watercourses (SLCs 35069 and 35064, which both fall in the highest total soil loss class, have the largest area within the watershed). On a t/ha basis, 11 SLCs representing 49% of row crop area and 45% of farmland, lost soil at a rate above 0.4 t/ha/yr and contributed some 60% of total 1996 soil losses in the basin. Three SLCs (35452, 350066 and 350074) fell in the highest t/ha soil loss class based on prevailing levels of row crops, tillage and soil clay content.

**Figure 6: Estimated soil loss per hectare into waters of the
Grand River basin, 1981, 1991, 1996**

Figure 7: Estimated total soil loss into waters of the Grand River basin, 1981, 1991, 1996

Figures 8 and 9, and Table A13, show changes in soil losses by SLC between 1981-1996. Total soil losses declined in 36 of 39 SLCs, and per hectare soil losses declined in all SLCs over this period, with 24 of these experiencing a decline of at least 20%.

The soil loss estimates reported here for the Grand River basin are within the range of values reported in other studies in southern Ontario (Table 31). Overall, these results suggest agriculture continues to contribute to sedimentation of waterways, but at a reduced rate compared to the early 1980s, particularly when reductions in tillage are taken into account. If only cropping factors are considered and all tillage is assumed to be conventional, soil losses into water are little changed between 1981, 1991 and 1996. The analysis also identifies those areas of the watershed on which the largest absolute (i.e. total tonnes) and relative (i.e. t/ha) soil losses occur, and which would thus benefit most from additional soil conservation measures.

**Change in tonnes of soil lost from cropland
into waters of the Grand River watershed by SLC:
1981 - 1996**

Reduction in tonnes of soil lost per hectare of cropland
into waters of the Grand River watershed by SLC:
1981 - 1996

Table 31 : Comparison of agricultural soil loss estimates from Southern Ontario			
Study		t/ha/yr)	Watershed(s)
Fox & Dickson, 1990	Conventional tillage	.55 - .65	Ranges are for all of Big Creek, Newbiggen Creek, Stratford/Avon watersheds
	Conservation tillage	.30 - .42	
	No-till	.13 - .16	
Wall et al, 1982	Conventional tillage	.077 - .24	Grand & sub-watersheds Saugeen & sub-watersheds
		.029 - .107	
This study	1981: all conventional tillage	.46	Mean values for Grand watershed
	1991: 79% conventional; 18% conservation; 3% no-till	.39	
	1996: 62% conventional; 22% conservation; 16% no-till	.34	

15.3 Economic damages from soil losses into water. Estimates of economic damages from soil losses into waters of the Grand River watershed under prevailing cropping and tillage practices are presented in Figure 10 and Tables A10, A11 and A14. All damage estimates are reported in constant 1996 dollars. Basin-wide damage estimates decreased by about 20%, from \$11.9 million in 1981 to \$9.5 million in 1996. On a dollar-per-hectare basis, average damage estimates decreased by 25%, from \$42.30 in 1981 to \$31.76 in 1996. Per hectare damage estimates in 1996 ranged from a high of \$59.00 in SLC 350452 to a low of \$7.61 in SLC 350076.

Figure 10: Economic damages from soil losses into waters of the Grand River basin under prevailing tillage practices, 1981, 1991, 1996.

The adoption of reduced tillage methods in the Grand River basin also resulted in considerable economic benefits when measured in terms of improved water quality (Figure 11 and Table A12). Annual basin-wide benefits of \$ 1.07 million in 1991 increased by about 119% to \$2.35 million in 1996. This sum represents about 25% of the total estimated economic damages imposed by the agriculture sector on other economic uses of water in the Grand River basin in 1996.

Table 32 also summarizes the results of the analysis of economic damages in 1996 across SLCs. The spatial distribution of economic damages to water quality from soil losses in 1996 is shown in Maps A3 and A4, and changes from 1981-1996 are shown in Table A14, and in Figures 12 and 13. Because a uniform \$/tonne of soil damage factor was applied throughout the watershed, the distribution of economic costs by SLC exactly mirrored the distribution of soil losses.

Figure 11: Economic benefits of reduced tillage to water quality in the Grand River basin, 1991, 1996.

Table 32: Distribution of total soil losses & economic costs in the Grand River basin across SLCs, 1996.										
Gross soil loss (tonnes)						Rate of soil loss (tonnes/ha)				
Range	No. of SLCs	% basin row crops	% basin soil loss	Soil loss (t/ha)	Damages \$ Million/(%)	Range	No. of SLCs	% basin row crops	% of basin soil loss	
< 1000	19	7	5	0.24	.42 / (4)	< 0.2	4	2	< 1	
1000 - 2000	5	10	6	0.27	.61 / (6)	.21 - .30	9	13	8	
2001 - 3000	7	17	16	0.39	1.51 / (16)	.31 - .40	15	36	32	
3001 - 6000	3	10	13	0.50	1.20 / (13)	.41 - .50	8	40	47	
6001 - 9000	3	20	21	0.43	2.00 / (21)	.51 - .65	3	9	13	
15000 - 26000	2	36	39	0.44	3.71 / (39)					

Change in total economic damages to water quality due to soil losses
from cropland into waters of the Grand River basin by SLC:
1981 - 1996

Reduction in economic damages to water quality per hectare due to soil losses from cropland into waters of the Grand River watershed by SLC:
1981 - 1996

16. Discussion and conclusion.

Agricultural land can affect welfare by providing a medium for crop growth and through the exchange of materials with other sectors and environmental media. This study has analyzed selected aspects of agricultural land's relationship to welfare in the Grand River watershed of Ontario using an environmental accounting approach. The stock of cropland soil in the basin is estimated in physical units, and flows of soil from cropland to water are estimated in both physical and economic units, for three accounting periods (1981, 1991, 1996). Summary results are portrayed in an environmental accounting framework in Tables 33 and 34 for the 1996 accounting period.

When viewed from an on-farm, net-balance perspective, soil stocks are estimated to be at virtual equilibrium, suggesting soil productivity (and producer utility) is not threatened by prevailing soil losses into water. Additional research would be required to determine whether gross erosion levels under prevailing management practices in 1996 pose a threat to soil productivity.

When results are viewed from a material-flow perspective, a different story emerges -- agricultural cropland loses (transfers) some 100 000 tonnes of soil to waters in the basin annually. Although this represents only a small proportion of the total available stock, this transfer nonetheless imposes an economic cost on water users of some \$9.5 million annually. Clearly, both stock and flow accounts are needed to provide a comprehensive picture of how agricultural soils affect welfare, and of how effects on welfare are distributed both on and off the farm.

Table 33: Balance sheet for cropland soil stocks in the Grand River basin		
Accounting period: 1996		
Soil Stock	Units	Quantity
Opening stock (15 cm depth)	Tonnes	640 244 525
Additions		332 927
Removals		103 470
Net change		229 457

Table 34: Flows of soil from cropland into waters of the Grand River basin					
Accounting period: 1996					
From:	Cropland	Watercourses		Total loss	
To:		Tonnes	1996 \$	Tonnes	1996 \$
Cropland	-----	103 470	9 482 021	103 460	NA
Watercourses	-----	-----	-----		9 482 021
Total gain		103 470	-----	-----	-----

The adoption by farmers of new, economically-viable conservation tillage technologies over the past 20 years has allowed them to successfully control soil losses from cropland into water below levels which threaten their economic health, but not below levels which fully protect socio-economic uses of water off the farm. These findings are consistent with the types of outcomes that might be expected given prevailing property right structures presently in place for soil and water in the Grand River basin. Because farmers enjoy strong and well-defined property rights over their

land, they have considerable latitude to determine which management practices to use (and not use), and an incentive to use and manage soils in a manner which preserves their income-generating potential over the longer term. In contrast, public property rights over water are weak and poorly defined relative to farmers' private property rights. Because not all resources are privately owned (i.e. water in this case), and not all costs (i.e. water pollution) resulting from the use of resources (i.e. land) accrue to the owners (i.e. farmers), the universality and exclusivity criteria for efficient property rights structures specified by Tietenberg (1996) are violated. Farmers therefore have little incentive to consider the off-site damage caused by soil losses into water, and an externality of crop production -- reduced water quality -- results.

The findings raise the question of which policy responses might be enacted to correct this form of market failure, and this in turn depends in part on how the on-farm costs of reducing soil losses into water compare with on-farm and off-farm benefits that result. Several studies have analysed the economics of land conservation practices and attempted to compare and contrast the value of the costs and benefits which result from their adoption in southern Ontario (Stonehouse et al., 1987; Stonehouse et al., 1988; Henderson and Stonehouse, 1988; Dickson and Fox, 1989; Weersink et al, 1992a, 1992b). The overall conclusion reached by these studies is that conservation practices are generally non-profitable to farmers, with the notable exception of conservation tillage, whose profitability varies widely according to such factors as climate, topography, soil physical properties, crop selection and farmer managerial skills (Stonehouse, 1996; Fox et al., 1995). Furthermore, many of these studies conclude that the off-farm benefits of

erosion control can outweigh the on-farm costs (Fox et al., 1995). For example, Fox and Dickson (1990) reported some instances where the off-farm benefits (to water quality) exceeded the on-farm costs of adopting conservation tillage practices, and other cases where they did not, depending on the watershed, tillage practice and cropping system used. Similarly, van Vuuren et al. (1997) reported a high, positive correlation between on-farm and off-farm benefits for some cropping and tillage practices in the Kettle Creek watershed. The adoption of reduced tillage techniques in the Grand River basin since 1981 suggests this practice has been economically viable for some farmers, and less so for others.

If both farmers and the public gain from the adoption of additional soil conservation measures, then there may be no role for public intervention except to make more farmers aware of the on-farm benefits of further adopting such measures. In cases where the public benefits from additional soil conservation measures exceed the costs to farmers of their adoption, public policy may have a legitimate role to play in encouraging the adoption of such practices. If property rights are to remain unchanged, public payments (i.e. subsidies) to farmers to offset the additional private costs incurred to change their practices represents one possible policy approach. An alternative approach would be to redefine property rights in favour of the public's right to cleaner water, leading to a weakening of farmers' existing property rights. The polluter-pays-principle might thus be more vigorously applied, farmers would be charged for the damage costs of soil losses into water, and thus provided with an economic incentive to reduce such losses to more efficient levels. However, additional study would be needed before firm policy

conclusions of this sort can be drawn. Examples of the types of information needed include the (on and off-farm) costs and benefits of various soil conservation practices, the most efficient level (value) of a subsidy or pollution charge, and information on the administrative feasibility and political costs of alternative policy options.

Tables 33 and 34 express the results of this analysis using a natural resource accounting approach (see Table 3). If a more aggregate approach were used as suggested by the Inter-Secretariat Working Group on National Accounts (1993) or Bartelmus et al., (1991), conventional economic aggregates (such as farm income or primary agricultural GDP) would be modified to take environmental externalities into account. Following this approach, the economic value of farm output in the Grand river basin would be adjusted downward to account for the costs of soil losses on water users by the amount estimated here. Put another way, as estimated in this study, the cost of producing agricultural products in the Grand River basin was greater than that captured through conventional accounting procedures by at least \$9.4 million in 1996.

Although this study has shed new light on linkages between the agricultural economy and the environment, the analysis provides but one example of the types of adjustments required to fully account for primary agriculture's relationship to overall welfare. For example, the productivity effects of changes in on-farm soil stocks might also be measured in economic terms, other elements of the accounting framework presented in Part C would be developed, and flows of other materials between the agricultural economy and the environment should also be accounted for, such as net emissions of greenhouse gases.

It is also clear that the results presented here are highly generalized and should be seen as providing first approximations only. Although the soil loss estimates are sensitive to variations in soil and agricultural factors across SLCs, they could be improved by taking additional factors that affect soil stocks and flows from cropland, such as riparian buffer strips, into account. There is also considerable scope for improving the economic valuation of damages to water quality. A single per-tonne damage cost estimate was applied across the study area, but such costs will likely vary within the basin. More precise data on the economic benefits accruing from improved water quality within the Grand River basin would improve the accuracy of the results.

Despite these limitations, this application suggests that environmental accounting has considerable potential for assessing the linkages between agricultural land and economic and ecological welfare, and for providing relevant information for policy and planning purposes. A key strength of the accounting approach is its ability to consider multiple dimensions of the relationship between agricultural land and overall welfare, including externalities of farm production, from both ecological and economic perspectives. This is evident from this study's consideration of soil stocks and flows both from the perspective of on-farm agricultural productivity, and from the perspective of users and other environmental media off the farm. Furthermore, there is considerable scope to account for other aspects of agricultural land's contribution to welfare, as suggested in the accounting framework developed in Part C of this report. Finally, although controversial, the expression of study results in economic terms allows for a more integrated comparison of the costs and benefits of agricultural production than is

possible in cases where these are expressed in different units. Such economic valuation also facilitates cost-benefit analysis of policy alternatives, and identification of those options most likely to maximize the contribution of the agricultural sector to overall (i.e. economic and ecological) welfare in the basin.

Part E: Overall Conclusions.

Perceptions about the meaning of sustainable development vary considerably, ranging from more ecocentric to more technocentric perspectives. Because a single and unifying theory of sustainable development has yet to emerge, analysis of its achievement in society remains difficult and elusive. Nonetheless, in light of continued increases in human populations and appropriation of natural capital by the socio-economy, there is overall consensus that indicators which take better account of linkages between economic activity and (social, economic and ecological) welfare are needed than presently available from conventional national accounts.

Work to this end is ongoing, and several frameworks for measuring progress toward (or away from) sustainable development have emerged, including stress-response frameworks, capital-based frameworks, input-output and theme-based frameworks. These are, to some extent, inter-related, and all have been used in a wide variety of applications, including for assessing sustainability in resource sectors such as agriculture.

The Environmental Accounting approach for analysing environment-economy relationships is a specific analytical approach to have recently come into use. This approach attempts to assess stocks and flows of natural capital in physical and/or economic units, and arose in response to concerns that conventional economic indicators derived from the national accounts were ignoring natural capital and thus sending misleading signals to policy-makers about relationships between economic activity and welfare.

This study has developed a conceptual framework for examining relationships

between agricultural land and welfare using an environmental accounting approach, and applied this approach to a specific aspect of primary agriculture in the Grand River watershed of Ontario. The results suggest environmental accounting holds promise as an approach for analysing linkages between agricultural land and welfare. If applied flexibly, through this approach land “stocks” can be defined to take various functions of land which affect economic and ecological welfare into account. Similarly, flows of various materials to and from agricultural lands, and their impacts on welfare in other resources and sectors, can be accounted for. This consideration of both stocks and flows of materials (i.e. of sources, stocks, destinations and any resulting economic/ecological impacts) allows for a more comprehensive analysis of environment - economy interactions and the identification of economy/environment linkages, and is a key advantage of environmental accounting.

This overall conclusion is supported by the more specific findings obtained through the application of an environmental accounting approach to stocks and flows of agricultural soil in the Grand River watershed of Ontario for the 1981, 1991 and 1996 accounting periods. The analysis shows that cropland soil stocks are at virtual equilibrium in the watershed, thus no substantial threat to farm productivity from soil losses into water is evident overall. However, from the perspective of public water users, the economic costs of soil losses (flows) are substantial. Clearly, by applying the environmental accounting approach, a more comprehensive picture emerges of how agricultural land affects welfare in this area than is available through conventional accounting procedures. A wider range of costs and benefits of agricultural production is

considered, and important insights are gained into how these compare with one another, and how they are distributed in society.

Information of this nature is highly relevant to policy. Results of the Grand River basin analysis show that strategies that aim to reduce soil losses into water would be most effective if targeted at areas of the watershed where the highest soil losses and economic damages occur. In cases where the public benefits of reducing soil losses exceed the private costs of additional conservation measures, policy might seek to compensate farmers for the additional costs of new measures, or, alternatively, apply the polluter-pays-principle by redefining existing property rights to favour the public's right to cleaner water.

One aspect of environmental accounting which differentiates it from some other environmental analytical frameworks (e.g. Stress-Response) is its inclusion of economic valuation and analysis. Although controversial, monetary valuation of the ecological costs and benefits of economic activity allows these to be weighed against other costs and benefits, possibly leading to more informed decisions about which policies are most likely to maximize welfare.

However, this study also underscores some of the limitations of environmental accounting. The results of the analysis in the Grand River basin, particularly the economic valuation of sedimentation damages, is highly sensitive to the estimated economic value of water uses in the basin, and the availability of reliable data on such uses. In this case the analysis could be improved with more detailed data of this nature. In general, the improvement of economic valuation methods so that they better recognize

and incorporate the full value of natural capital (i.e. environmental goods and services) to human welfare would help address a key criticism of this aspect of environmental accounting.

Related to this is the question of how information obtained through environmental accounting analysis is to be interpreted from a sustainability perspective. There are no clear answers here given the lack of consensus about the meaning of sustainable development. Perhaps it is sufficient to note that if environmental accounting analyses are expressed in physical terms, to the extent that they can be compared with established ecological limits and thresholds (e.g. water quality guidelines for contaminants, or pollution release limits) then some level of interpretation from a “strong sustainability” perspective may be possible. If expressed in economic terms, interpretation can consider how changes in natural capital compare with changes in other forms of capital, which facilitates interpretation from a “weak sustainability” perspective.

Potential next steps for this research might involve applying and evaluating other aspects of the accounting framework developed in Part C for other functions of agricultural land, as well as for agricultural land use and land management. A fully-developed set of accounts as proposed here would provide a more comprehensive overview of how agricultural lands affect economic and ecological welfare, and whether their use and management is such that their capacity to contribute positively to welfare is being maintained. Finally, a still broader set of accounts, particularly of flow accounts for nutrients, energy and other materials, would improve understanding of how the primary agricultural economy affects overall welfare.

At the national level, recent work on analysis of the relationship between primary agriculture and the environment is reported in McRae et al., (2000). That study did not employ an accounting approach, but instead used a Driving Force-Outcome-Response framework (a variation of the Pressure-State-Response model), and expressed results in physical units only. Nonetheless, the findings offer a starting point for applying an environmental accounting approach to the analysis of primary agriculture and the environment in Canada. Economic research to value results in economic terms represents another potential direction of future research that could yield additional insights into how Canadian agriculture affects national and even international welfare.

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Appendix A: Detailed results of soil loss and economic damage calculations for the Grand River watershed.

Soil loss from cropland into waters of the Grand River basin by SLC:
tonnes, 1996

Soil loss from cropland into waters of the Grand River basin by SLC:
tonnes per hectare, 1996

**Economic damages to water quality due to soil losses from cropland
into waters of the Grand River watershed by SLC: 1996**

Economic damages to water quality due to soil losses from cropland into waters of the Grand River watershed by SLC: 1996

Table A1: Soil Landscape Polygons lying wholly or partially within the Grand River basin.

SLC No.	Ha. in Grand River basin	% in Grand River basin	Total area (ha)
350031	2281	5	48851
350032	9155	25	36283
350033	67	< 1	18078
350035	7	< 1	44598
350036	12516	100	12518
350045	57436	80	71432
350047	33054	13	249932
350048	24469	65	37907
350049	60396	100	60396
350050	6723	26	25716
350051	837	82	1024
350052	793	3	23541
350062	527	< 1	188856
350064	90979	37	243579
350066	16607	100	16607
350067	26372	100	26372
350068	14269	100	14269
350069	84526	99	85288
350070	38436	64	59936
350071	7683	34	22345
350072	17990	43	42269
350073	27020	100	27020
350074	19508	55	35620
350076	22	< 1	14376
350077	8890	79	11307
350078	22430	100	22430
350079	7962	33	24229
350080	38380	33	116784
350144	11397	43	26729
350145	4105	44	9315
350147	589	1	54338
350148	397	4	9721
350153	8962	76	11801
350426	2313	31	7533
350429	2468	56	4426
350432	726	51	1414
350452	6134	100	6134
350470	11395	54	20971
350471	12988	94	13836
Total	690809	40	1747781

Source: Compiled from Canadian Soil Information System (CANSIS), Agriculture and Agri-Food Canada.

Table A2: Area (ha) of farmland and row crops by Soil Landscape Polygon, 1981, 1991, 1996 (includes area within and outside of Grand river basin)

SLC	81 Farmland	91 Farmland	96 Farmland	81 Row Crops	91 Row Crops	96 Row Crops
350031	40317	38384	39809	28448	26252	27260
350032	26840	26598	30345	18251	17850	21167
350033	15230	12576	13928	9503	8253	9749
350035	46096	38532	40145	30662	26177	28708
350036	7440	11597	10199	4918	7556	6765
350045	43266	41383	42245	26337	23133	23846
350047	146561	125560	134505	72095	48891	59911
350048	28889	21048	23600	14636	8609	10516
350049	35864	34868	33963	21441	21054	21451
350050	17775	18299	19879	12584	12645	13581
350051	413	388	446	221	181	193
350052	13067	12086	15278	7495	6411	8725
350062	160910	155741	160672	89722	92518	101214
350064	202542	198410	210440	130854	131045	142602
350066	11923	14548	15122	6617	7509	7863
350067	15720	11794	12521	10020	7121	7657
350068	7436	8280	7959	4211	4886	4264
350069	68164	66657	67145	37622	38068	40009
350070	30528	28459	26065	11631	10198	10236
350071	8618	13646	11374	3120	5262	4653
350072	24736	19505	20964	9422	6417	7240
350073	20157	17660	21058	9869	7773	10848
350074	30998	31501	33390	16684	16391	18943
350076	6640	5544	7804	1979	1392	1565
350077	7409	7258	6627	2345	2147	2184
350078	13468	13747	12310	4963	4307	3923
350079	13980	13450	15093	5857	4412	4534
350080	75051	67263	65589	25076	19721	19758
350144	17220	16340	15787	9046	8053	8365
350145	5046	5073	6082	2111	2229	2527
350147	13770	8969	12341	6233	3952	6317
350148	4117	5532	3769	1402	1994	1382
350153	8265	6413	6437	3435	2129	2332
350426	7823	5868	6644	5687	3927	4651
350429	3044	2613	2976	1560	808	1072
350432	795	568	769	446	186	325
350452	6204	6864	6332	3629	4427	3813
350470	19843	14909	16557	14306	9944	11028
350471	12384	12000	10336	7920	8168	6744
Total	1218549	1139931	1186505	672358	611996	667921

Source: Compiled from the Canadian Soil Information System (CANSIS), Agriculture and Agri-Food Canada.

Table A3: Soil clay content and distribution of row crops within Soil Landscape Polygons (includes area within and outside of Grand river basin).

SLC	% Clay	% Farmland in row crop: 1981	% Farmland in row crop: 1991	% Farmland in row crop: 1996
350031	11	71	68	68
350032	29	68	67	70
350033	19	62	66	70
350035	17	67	68	72
350036	14	66	65	66
350045	38	61	56	56
350047	43	49	39	45
350048	47	51	41	45
350049	13	60	60	63
350050	11	71	69	68
350051	43	53	46	43
350052	24	57	53	57
350062	18	56	59	63
350064	23	65	66	68
350066	37	55	52	52
350067	11	64	60	61
350068	13	57	59	54
350069	17	55	57	60
350070	19	38	36	39
350071	19	36	39	41
350072	11	38	33	35
350073	32	49	44	52
350074	39	54	52	57
350076	12	30	25	20
350077	18	32	30	33
350078	19	37	31	32
350079	14	42	33	30
350080	18	33	29	30
350144	12	53	49	53
350145	15	42	44	42
350147	33	45	44	51
350148	15	34	36	37
350153	18	42	33	36
350426	12	73	67	70
350429	47	51	31	36
350432	34	56	33	42
350452	39	58	64	60
350470	13	72	67	67
350471	11	64	68	65
Mean	23	53	50	52

Source: Canadian Soil Information System, Agriculture and Agri-Food Canada.

Table A4: Area (ha) of row crops under conventional, conservation and no-till tillage by SLC: 1991 & 1996 (includes area within and outside of Grand River Basin).

SLC	Conventional 91	Conservation 91	No-till 91	Conventional 96	Conservation 96	No-till 96
350031	16542	8702	1008	16413	6949	3898
350032	13156	4232	462	11828	5428	3910
350033	6160	1968	125	5407	2349	1993
350035	21028	4807	343	18285	5707	4716
350036	5231	2178	147	3464	2408	893
350045	17037	4798	1298	11749	5568	6529
350047	41323	6162	1405	31912	10247	17752
350048	6852	1282	474	4738	1118	4660
350049	13941	6356	757	9987	7883	3581
350050	7850	4089	706	8850	3062	1669
350051	150	22	9	87	28	79
350052	5134	1049	228	3534	2373	2818
350062	78933	10785	2799	64461	18370	18383
350064	113274	15362	2409	98286	23681	20635
350066	6723	741	45	6559	1024	280
350067	4592	2229	300	3417	2907	1334
350068	4296	539	50	3706	451	107
350069	28928	8315	825	25997	11078	2934
350070	8280	1610	308	5416	2507	2314
350071	2478	2258	525	2270	1911	473
350072	4940	1393	83	4255	2821	164
350073	6575	1168	31	7707	3007	133
350074	13959	2314	118	15184	3099	660
350076	1115	226	50	1341	224	0
350077	1571	439	137	1565	620	0
350078	3126	1040	141	2580	1311	32
350079	3963	401	48	3526	980	27
350080	16266	3167	288	16125	3571	62
350144	6159	1719	174	5133	2596	637
350145	1919	268	42	1623	678	227
350147	2748	1117	87	2614	1435	2269
350148	1405	186	402	529	669	184
350153	1630	478	21	1573	751	7
350426	3435	404	89	2750	1163	739
350429	657	82	69	452	142	478
350432	167	16	3	175	46	104
350452	3009	1360	57	3139	614	61
350470	6272	3206	467	5409	3350	2270
350471	6008	2043	117	3249	2513	983
Total	486832	108511	16647	415295	144639	107995

Source: Compiled from the Canadian Soil Information System, Agriculture and Agri-Food Canada.

Table A5: Estimated per-hectare soil loss into water by Soil Landscape Polygon assuming 100% conventional tillage (includes area within and outside of Grand river basin).

SLC	Kg/ha 1981	Kg/ha 1991	Kg/ha 1996
350031	478	454	454
350032	653	645	669
350033	494	525	557
350035	508	516	547
350036	469	461	469
350045	695	655	655
350047	657	578	626
350048	717	638	669
350049	417	417	441
350050	478	462	454
350051	689	634	610
350052	505	473	505
350062	442	465	497
350064	565	573	588
350066	635	612	612
350067	418	386	394
350068	389	405	366
350069	420	436	459
350070	305	289	313
350071	289	313	329
350072	217	178	194
350073	536	496	560
350074	650	634	674
350076	167	128	88
350077	246	230	253
350078	297	250	258
350079	280	209	185
350080	260	228	236
350144	342	311	342
350145	291	306	291
350147	515	507	562
350148	227	243	251
350153	324	253	276
350426	509	462	485
350429	716	558	597
350432	612	431	502
350452	682	729	698
350470	512	473	473
350471	423	454	431
Mean	462	436	451

Source: Author's calculations.

Table A6: Estimated soil loss into waters of the Grand River basin assuming 100% conventional tillage, tonnes, 1981, 1991, 1996.

SLC	1981	1991	1996
350031	680	596	651
350032	2980	2879	3456
350033	17	16	18
350035	2	2	2
350036	2308	3486	3174
350045	14639	12127	13255
350047	6159	3675	5119
350048	6818	3568	4899
350049	8950	8788	8954
350050	1564	1519	1688
350051	125	94	109
350052	114	91	132
350062	111	120	125
350064	27341	27763	29795
350066	4204	4593	4995
350067	4187	2751	3200
350068	1639	1979	1660
350069	15632	16413	16624
350070	2272	1889	1999
350071	307	560	458
350072	880	490	676
350073	5290	3859	5814
350074	5967	5720	6775
350076	0	0	0
350077	455	390	424
350078	1476	1076	1166
350079	540	304	418
350080	2150	1486	1694
350144	1331	1076	1231
350145	270	300	323
350147	32	20	33
350148	13	19	13
350153	845	409	574
350426	898	562	734
350429	626	252	430
350432	139	41	102
350452	2475	3229	2600
350470	3957	2538	3050
350471	3146	3487	2679
Total	130539	118167	129049

Source: Author's calculations.

Table A7: Estimated soil loss into waters of the Grand River basin under prevailing practices, tonnes, 1981, 1991, 1996.

SLC	1981	1991 Change (%), 1981-91		1996 Change (%), 1991-96 Change (%), 1981-96		
350031	680	498	-27	486	-2	-29
350032	2980	2544	-15	2665	5	-11
350033	17	14	-18	15	7	-12
350035	2	2	0	2	0	0
350036	2308	3024	31	2390	-21	4
350045	14639	10577	-28	8667	-18	-41
350047	6159	3405	-45	3417	0	-45
350048	6818	3200	-53	2810	-12	-59
350049	8950	7464	-17	6826	-9	-24
350050	1564	1254	-20	1304	4	-17
350051	125	85	-32	60	-29	-52
350052	114	82	-28	85	4	-25
350062	111	112	1	110	-2	-1
350064	27341	26044	-5	25475	-2	-7
350066	4204	4387	4	4422	1	5
350067	4187	2311	-45	2146	-7	-49
350068	1639	1875	14	1462	-22	-11
350069	15632	14679	-6	15108	3	-3
350070	2272	1723	-24	1488	-14	-35
350071	307	419	36	393	-6	28
350072	880	442	-50	496	12	-44
350073	5290	3611	-32	5329	48	1
350074	5967	5359	-10	6365	19	7
350076	0	0	#DIV/0!	0	#DIV/0!	#DIV/0!
350077	455	338	-26	387	14	-15
350078	1476	943	-36	867	-8	-41
350079	540	290	-46	251	-13	-54
350080	2150	1372	-36	1423	4	-34
350144	1331	964	-28	1003	4	-25
350145	270	281	4	265	-6	-2
350147	32	17	-47	22	29	-31
350148	13	16	23	10	-38	-23
350153	845	368	-56	424	15	-50
350426	898	529	-41	543	3	-40
350429	626	225	-64	216	-4	-65
350432	139	39	-72	58	49	-58
350452	2475	2793	13	2453	-12	-1
350470	3957	2113	-47	2019	-4	-49
350471	3146	3094	-2	2008	-35	-36
Total	130539	106493	-18	103470	-3	-21

Note: Soil loss from SLC 350076 < 0.01 tonnes.

Source: Author's calculations.

Table A8: Estimated per-hectare soil loss into waters of the Grand River basin under prevailing practices, tonnes, 1981, 1991, 1996.

SLC	1981	1991	1996
350031	0.48	0.38	0.36
350032	0.65	0.57	0.50
350033	0.49	0.47	0.41
350035	0.51	0.47	0.43
350036	0.47	0.40	0.35
350045	0.69	0.57	0.45
350047	0.66	0.54	0.44
350048	0.72	0.57	0.41
350049	0.42	0.35	0.32
350050	0.48	0.38	0.37
350051	0.69	0.58	0.38
350052	0.50	0.43	0.32
350062	0.44	0.43	0.39
350064	0.56	0.54	0.48
350066	0.64	0.58	0.56
350067	0.42	0.32	0.28
350068	0.39	0.38	0.34
350069	0.42	0.39	0.38
350070	0.31	0.26	0.23
350071	0.29	0.23	0.25
350072	0.22	0.16	0.16
350073	0.54	0.46	0.49
350074	0.65	0.59	0.61
350076	0.17	0.12	0.08
350077	0.25	0.20	0.22
350078	0.30	0.22	0.22
350079	0.28	0.20	0.17
350080	0.26	0.21	0.22
350144	0.34	0.28	0.28
350145	0.29	0.29	0.24
350147	0.51	0.44	0.35
350148	0.23	0.20	0.18
350153	0.32	0.23	0.24
350426	0.51	0.43	0.38
350429	0.72	0.50	0.36
350432	0.61	0.41	0.35
350452	0.68	0.63	0.64
350470	0.51	0.39	0.34
350471	0.42	0.40	0.32
Mean	0.46	0.39	0.35

Source: Author's calculations.

Table A9: Effect of reduced tillage on estimated soil losses into waters of the Grand River basin, tonnes, 1991, 1996.

SLC	Conventional tillage 1991	Reduced tillage 1991	% Change	Conventional tillage 1996	Reduced tillage 1996	% Change
350031	596	498	-16	651	486	-25
350032	2879	2544	-12	3456	2665	-23
350033	16	14	-13	18	15	-17
350035	2	2	0	2	2	0
350036	3486	3024	-13	3174	2390	-25
350045	12127	10577	-13	13255	8667	-35
350047	3675	3405	-7	5119	3417	-33
350048	3568	3200	-10	4899	2810	-43
350049	8788	7464	-15	8954	6826	-24
350050	1519	1254	-17	1688	1304	-23
350051	94	85	-10	109	60	-45
350052	91	82	-10	132	85	-36
350062	120	112	-7	125	110	-12
350064	27763	26044	-6	29795	25475	-14
350066	4593	4387	-4	4995	4422	-11
350067	2751	2311	-16	3200	2146	-33
350068	1979	1875	-5	1660	1462	-12
350069	16413	14679	-11	16624	15108	-9
350070	1889	1723	-9	1999	1488	-26
350071	560	419	-25	458	393	-14
350072	490	442	-10	676	496	-27
350073	3859	3611	-6	5814	5329	-8
350074	5720	5359	-6	6775	6365	-6
350076	0	0	#DIV/0!	0	0	#DIV/0!
350077	390	338	-13	424	387	-9
350078	1076	943	-12	1166	867	-26
350079	304	290	-5	418	251	-40
350080	1486	1372	-8	1694	1423	-16
350144	1076	964	-10	1231	1003	-19
350145	300	281	-6	323	265	-18
350147	20	17	-15	33	22	-33
350148	19	16	-16	13	10	-23
350153	409	368	-10	574	424	-26
350426	562	529	-6	734	543	-26
350429	252	225	-11	430	216	-50
350432	41	39	-5	102	58	-43
350452	3229	2793	-14	2600	2453	-6
350470	2538	2113	-17	3050	2019	-34
350471	3487	3094	-11	2679	2008	-25
Total	118167	106493	-10	129049	103470	-20

Source: Author's calculations.

Table A10: Total damage costs in the Grand river basin from soil losses into water under prevailing practices: 1981, 1991, 1996 (1996 dollars).

SLC	1981	1991	1996
350031	62294	45642	44576
350032	273076	233102	244186
350033	1583	1304	1363
350035	224	177	179
350036	211463	277132	219065
350045	1341520	969306	794271
350047	564454	312012	313157
350048	624792	293251	257489
350049	820135	683977	625521
350050	143290	114892	119523
350051	11427	7831	5540
350052	10402	7556	7745
350062	10129	10226	10102
350064	2505486	2386699	2334497
350066	385213	402011	405199
350067	383710	211739	196623
350068	150217	171787	133948
350069	1432517	1345176	1384485
350070	208195	157928	136388
350071	28136	38404	35971
350072	80638	40512	45480
350073	484744	330951	488319
350074	546815	491136	583320
350076	0	0	0
350077	41684	30986	35459
350078	135224	86449	79484
350079	49521	26555	22989
350080	197069	125702	130390
350144	122018	88377	91944
350145	24739	25787	24324
350147	2939	1592	2049
350148	1169	1433	890
350153	77426	33736	38869
350426	82255	48457	49740
350429	57325	20648	19777
350432	12761	3566	5293
350452	226781	255908	224821
350470	362602	193660	185039
350471	288299	283490	184008
Total Cost	11962272	9759097	9482023

Source: Author's calculations.

Table A11: Per hectare damage costs from soil losses into waters of the Grand River basin under prevailing practices: 1981, 1991, 1996 (1996 dollars).

SLC	1981	1991	1996
350031	44	35	33
350032	60	52	46
350033	45	43	38
350035	47	43	40
350036	43	37	32
350045	64	52	42
350047	60	49	40
350048	66	52	38
350049	36	32	29
350050	44	35	34
350051	63	53	35
350052	46	39	30
350062	39	40	36
350064	52	49	44
350066	58	54	52
350067	38	30	26
350068	31	35	31
350069	38	36	35
350070	28	24	21
350071	27	21	23
350072	20	15	15
350073	49	43	45
350074	60	54	56
350076	15	11	8
350077	23	18	21
350078	27	20	20
350079	26	18	15
350080	24	19	20
350144	31	26	26
350145	27	26	22
350147	47	40	32
350148	21	18	16
350153	30	21	22
350426	47	40	34
350429	66	46	33
350432	56	38	32
350452	62	58	59
350470	47	36	31
350471	38	37	29
Mean	42	36	32

Source: Author's calculations.

Table A12: Soil loss reduction benefits of reduced tillage in the Grand river basin, 1991 and 1996 (all monetary values in 1996 \$).

SLC	1991 (tonnes)	1991 (\$)	1996 (tonnes)	1996 (\$)
350031	98	8992	165	15117
350032	335	30744	791	72516
350033	2	159	3	261
350035	0	16	0	31
350036	461	42282	784	71810
350045	1550	142022	4587	420394
350047	270	24758	1701	155907
350048	368	33750	2089	191450
350049	1324	121334	2128	194988
350050	266	24331	383	35120
350051	8	763	49	4469
350052	9	785	48	4363
350062	9	780	14	1324
350064	1719	157543	4320	395920
350066	206	18870	573	52551
350067	440	40337	1054	96608
350068	105	9591	198	18177
350069	1734	158913	1516	138939
350070	165	15162	511	46838
350071	141	12927	65	5988
350072	48	4422	180	16484
350073	248	22702	486	44502
350074	360	33032	410	37534
350076	0	0	0	0
350077	52	4737	37	3379
350078	133	12188	299	27393
350079	14	1263	167	15346
350080	114	10432	271	24880
350144	111	10214	228	20891
350145	19	1749	58	5283
350147	3	242	10	929
350148	4	345	3	262
350153	40	3700	149	13690
350426	33	3054	191	17529
350429	27	2483	214	19622
350432	2	176	44	4010
350452	436	39970	147	13458
350470	425	38957	1031	94493
350471	394	36080	671	61504
Total	11673	1069805	25575	2343960

Source: Author's calculations.

Table A13: Change in soil loss into waters of the Grand river basin between 1981 and 1996.

SLC	Soil loss class (tonnes)	Soil loss class (t/ha)
350031	3	3
350032	4	3
350033	4	2
350035	4	2
350036	5	3
350045	2	4
350047	2	4
350048	1	5
350049	3	3
350050	4	3
350051	1	5
350052	3	4
350062	5	2
350064	4	2
350066	6	2
350067	2	4
350068	4	2
350069	5	1
350070	3	3
350071	7	2
350072	2	3
350073	5	1
350074	6	1
350076	1	5
350077	4	1
350078	2	3
350079	1	5
350080	3	2
350144	3	2
350145	5	2
350147	3	4
350148	3	3
350153	2	3
350426	2	3
350429	1	5
350432	1	5
350452	5	1
350470	2	4
350471	2	3

Tonnes: 1 = reduction of > 50%; 2 = reduction of 36 - 50%; 3 = reduction of 21 < 36%; 4 = reduction of 5 < 21%; 5 = no change (+/- 5%); 6 = increase of 5 < 20%; 7 = increase of 20 - 35%.

Tonnes/ha: 1 = reduction of 0 < 10%; 2 = reduction of 10 < 20%
3 = reduction of 20 < 30%; 4 = reduction of 30 < 40%; 5 = reduction of 40-50%.

Table A14: Change in economic damages to water quality in Grand river basin between 1981 and 1996 (based on 1996 dollar values).

SLC	Class (total damages)	Class (damages / ha)
350031	3	3
350032	4	3
350033	4	2
350035	4	2
350036	5	3
350045	2	4
350047	2	4
350048	1	5
350049	3	3
350050	4	3
350051	1	5
350052	3	4
350062	5	2
350064	4	2
350066	6	2
350067	2	4
350068	4	2
350069	5	1
350070	3	3
350071	7	2
350072	2	3
350073	5	1
350074	6	1
350076	1	5
350077	4	1
350078	2	3
350079	1	5
350080	3	2
350144	3	2
350145	5	2
350147	3	4
350148	3	3
350153	2	3
350426	2	3
350429	1	5
350432	1	5
350452	5	1
350470	2	4
350471	2	3

Total damages: 1 = reduction of > 50%; 2 = reduction of 36 - 50%; 3 = reduction of 21 < 36%; 4 = reduction of 5 < 21%; 5 = no change (+/- 5%); 6 = increase of 5 < 20%; 7 = increase of 20 - 35%.

Damages/ha: 1 = reduction of 0 < 10%; 2 = reduction of 10 < 20%; 3 = reduction of 20 < 30%; 4 = reduction of 30 < 40%; 5 = reduction of 40-50%.